THE 30-YEAR OUTCOME OF ASSISTED REGENERATION TREATMENTS

IN A BURNED AND SALVAGED INTERIOR ALASKA BOREAL FOREST

By

Andrew Allaby

RECOMMENDED:

Dr. John Yarie

Dr. Brian Young

Dr. Glenn Juday
Advisory Committee Chair

Dr. Peter Fix
Chair, Department of Natural Resources Management

APPROVED:

Dr. David Valentine
Director of Academic Programs
School of Natural Resources and Extension

Dr. John Eichelberger
Dean of the Graduate School

24 August 2015
Date

THE 30-YEAR OUTCOME OF ASSISTED REGENERATION TREATMENTS IN A BURNED AND SALVAGED INTERIOR ALASKA BOREAL FOREST

A

THESIS

Presented to the Faculty

of the University of Alaska Fairbanks

in Partial Fulfillment of the Requirements

for the Degree of

MASTER OF SCIENCE

By

Andrew Allaby, B.A.

Fairbanks, AK

August 2015

Abstract

This study contributes to the understanding of the persistence of silvicultural treatments into the stem exclusion stage of forest development in an experiment originally designed to test the effectiveness of various white spruce (*Picea glauca* Moench [Voss]) regeneration practices. Many studies in the North American boreal forest address the effect of silvicultural treatments on a single tree species, specifically white spruce in the great majority of cases. The experiment measured in this study provided an excellent opportunity to compare treatment effects on white spruce density and growth. The Rosie Creek Fire Tree Regeneration Installation experiment represents an operational-scale, spatially-explicit, replicated design on a single site disturbed consecutively by high-severity wildfire and clearcut salvage harvest. Three hierarchical factors, each with multiple levels, were examined: landform type, ground scarification methods, and white spruce regeneration methods. All three of the experimental factors exercised continuing influence on the patterns of white spruce regeneration and growth. The treatment effects did not attenuate over time for white spruce, and we found statistically significant effects that the original researchers could only describe as tendencies.

However, relatively few studies address treatment impacts on non-target species or determine how the silvicultural treatments affect a site's overall woody biomass production. Experimental silvicultural practices targeted in this study to improve white spruce survival had profound effects on other dominant upland tree species such as quaking aspen (*Populus tremuloides* Michx.) and Alaska birch (*Betula neoalaskana* Sarg.). Interior Alaska timber species demonstrate different regeneration strategies to post-disturbance environmental conditions, especially residual organic soil layer thickness and spatial configuration of surviving potential seed sources. Effective silvicultural practices must consider each species' unique reproductive biology, and clonal sprouting as a source of aspen persistence was a particularly important example in our study. Site differences, such as we found between the slope and ridge landforms, are a key consideration for implementing effective silvicultural practices. Significant interactions between the regeneration treatments and landform types proved to be critical to meet specific reforestation objectives, particularly the different herbaceous vegetation cover types, presence/absence of aspen clonal rootstocks, and spatial configurations regarding seed sources. Managing mixed species stands, which are common in the lightly managed portions of the boreal forest, requires not only the consideration of the future crop tree, but the interacting effects of silvicultural practices on all tree species.

Table of Contents

List of Figures

Page

List of Tables

Page

List of Appendices

Page

Acknowledgements

To my graduate committee for their interest in, commitment to, and tireless advocacy of my project and career; my wife Eva for her unflagging support; Professor Julie McIntyre for guiding me through the shoals of statistics; and the dedicated technicians – particularly Kristy Johnsson and Ryan Jess – who tirelessly measured thousands of trees: thank you! None of this research would have been possible without generous and ongoing support from the Boreal Alaska–Learning, Adaptation, and Production partnership, the McIntire-Stennis program, and Bonanza Creek LTER. And lastly, I would like to acknowledge a debt of gratitude to Roseann V. Densmore, John C. Zasada, and Glenn P. Juday, who with foresight established the Rosie Creek Fire Tree Regeneration Installation to the benefit of future Alaskans.

Chapter 1 – General Introduction: A review of forest management for burned and salvaged sites

1.1 Introduction

Modern forest management seeks to equal or enhance the value stream that flows from intact natural ecosystems, whether those values are physical products, ecosystem services, or policy goals such as sustaining biodiversity (Puettmann *et al.*, 2009). The strategic goals of forest management are ultimately achieved at the stand level (i.e., at the operational scale of typical management activities) through the application of silvicultural practices (Kneeshaw *et al.*, 2000). Silvicultural practices are those activities that "control the establishment, composition, structure, growth, and role of trees within managed forests" (Puettmann *et al.*, 2009). Silvicultural practices target key characteristics such as regeneration, tree growth and vigor, species diversity, and structural complexity, which are correlated with the capacity of an ecosystem to provide multiple values in perpetuity (Seymour and Hunter, 1999). However, if the intended effects of various silvicultural practices involve a high degree of uncertainty, the ability to achieve various forest management goals efficiently may be diminished (Ogden and Innes, 2007).

One way to optimize multiple management goals is through the implementation of ecological forestry, in which the full range of ecosystem patterns and processes is maintained over time (Seymour and Hunter, 1999; McRae *et al.*, 2001; Puettmann *et al.*, 2009). This holistic approach, often referred to as ecosystem management or natural disturbance-based management, commits managers to maintaining the integrity of forest ecosystems and processes (Kimmins, 2004). Healthy forest ecosystems are those with unimpaired productivity, species diversity, nutrient cycling, population cycles, and successional processes (Kolb *et al.*, 1994). In order to maintain a healthy forest, the central principle of ecological forestry directs managers to consciously reproduce the natural disturbance regime and associated impacts on forest characteristics (Seymour and Hunter, 1999; Drever *et al.*, 2006). Pickett and White (1985) define a disturbance as "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability or the physical environment." A disturbance regime characterizes the frequency, severity, and spatial pattern of disturbances within an ecosystem. Disturbance regimes represent a selective pressure to which plants evolve, community associations form, and ecosystem processes develop (Weber and Flannigan, 1997). Ecological forestry strives to consciously reproduce the conditions that follow typical disturbances, particularly biological legacies, competitive tree mortality processes, and sufficient recovery periods to develop structural and biological complexity (Franklin *et al.*, 2007).

Emulating the natural disturbance regime when the regime itself is changing quickly in response to global climate change is becoming increasingly difficult. Current global temperatures are likely the highest they have been in 1,400 years, and atmospheric concentrations of greenhouse gases like carbon dioxide (CO_2) and methane (CH_4) are likely the highest for the past 800,000 years (Blanco *et al.*, 2014). The Arctic is warming more quickly than the global average (Juday *et al.*, 2005; Blanco *et al.*, 2014), and the boreal forest is undergoing unprecedented changes in plant range shifts, disturbance regimes, and insect outbreaks (Soja *et al.*, 2007; Johnstone *et al.*, 2010a). Climate dynamics directly affect disturbance regimes, with increasing heat and variable precipitation as particularly strong influences on the wildfire regimes in fire-adapted ecosystems (Weber and Flannigan, 1997).

Frequent, stand-replacing wildfire is the principal disturbance to the boreal forest (Heinselman, 1981; Kasischke *et al.*, 2006). A disturbance regime is described by its typical frequency, severity, and extent (Pickett and White, 1985). As a consequence of climate change, the historical fire regimes are currently shifting toward greater area burned and shorter return intervals across not only Alaska but many of North America's boreal forest regions (Weber and Flannigan, 1997; Kasischke *et al.*, 1999; Kasischke and Turetsky, 2006; Kasischke *et al.*, 2010). After a fire, most future canopy tree species recruit within a relatively short window of time, facilitating re-establishment of the dominant pre-fire species (Chapin *et al.*, 2004). However, increasing fire severity and frequency reduces the capacity of an ecosystem to return to its pre-disturbance state (Johnstone and Chapin, 2006; Johnstone *et al.*, 2010b; Johnstone *et al.*, 2011).

A changing disturbance regime not only affects the proportion of a landscape in various seral stages, it may also change a site's successional pathway by reducing the fitness of some species. For example, very short fire return intervals reduce conifer regeneration in favor of light-seeded hardwood species (Johnstone *et al.*, 2010b). The unusually severe fires of recent decades also appear to alter post-fire community composition by combusting a greater quantity of forest floor and residual seed sources, reinforcing the advantage of light-seeded hardwoods (Johnstone and Chapin, 2006). A warming climate may additionally cause drought- or heat-stressed trees to experience slower growth, less vigor to resist insects and disease, and less reproductive success (Barber *et al.*, 2000; Juday *et al.*, 2003; Beck *et al.*, 2011), which renders less certain the capacity of management activities to influence forest characteristics.

The intensifying wildfire regime, coupled with other climate change perturbations such as insect outbreaks, poses particular challenges to the maintenance of productive ecosystems over long time periods (Lemmen and Warren, 2004). Silvicultural practices in these fire-prone ecosystems typically

emulate post-fire conditions. For example, irregularly-shaped clearcuts approximate the large patches of even-aged forest that regenerate after a severe fire (Bergeron *et al.*, 2002; Ilisson and Chen, 2009). However, when typical post-disturbance regeneration conditions are fundamentally altered, previously satisfactory practices may no longer produce acceptable tree recruitment and growth rates to meet human needs. Uncertainty about regional climate change effects and feedbacks increases the difficulty to produce desired forest states through management actions. Climate models downscaled to levels useful to forest planning often contain unworkably broad confidence intervals and caveats (Ogden and Innes, 2009), and positive and negative feedbacks interacting across multiple temporal and spatial scales add to this uncertainty (Ohlson *et al.*, 2005). Forest management during a rapidly changing climate, whether based on ecological forestry or some other paradigm, must explicitly incorporate these future uncertainties (Ogden and Innes, 2007).

Management responses to climate change may be categorized as either mitigation activities, which aim to reduce climate change, or adaptation activities, which may improve the resistance, resilience, and response of an ecosystem to the implications of climate change (Swanston *et al.*, 2012). Silvicultural practices can be designed to enhance both society's and an ecosystem's ability to adapt to climate change, particularly by conserving a forest's productive potential by building resilience (Ogden and Innes, 2007). Ecological resilience is the capacity of a forest to, "not only accommodate gradual changes related to climate but tend to return toward a prior condition after disturbance either naturally or with management assistance" (Millar *et al.*, 2007). A resilient forest that maintains ecosystem processes is more likely to provide social benefits over time (Ogden and Innes, 2007). The capacity of forest management to enhance resilience rests on effective operational-scale silvicultural practices, specifically practices that quickly return a forest to its pre-disturbance state in accordance with ecological forestry. It should be noted that climate change may exceed the adaptive capacity of existing tree species (Barber *et al.*, 2000; Juday *et al.*, 2003; Beck *et al.*, 2011), and maintaining ecosystem resilience may eventually exceed the reach of operational scale practices.

Two complementary types of silvicultural practices with particular relevance to climate change adaptation amid an accelerating wildfire regime are salvage harvest of burned forests and assisted regeneration. Salvage harvest takes place within a few years after a fire, and recovers value from dead standing timber that would otherwise degrade over time (Spittlehouse and Stewart, 2003; Bergeron *et al.*, 2004; Lemmen and Warren, 2004; Le Goff *et al.*, 2005; Millar *et al.*, 2007). Where unprecedented portions of the forest resource are affected by fire (Calef *et al.*, 2015), salvage harvest mitigates socio-economic dislocations to forest-dependent communities by capturing residual value of dead timber

(Spittlehouse and Stewart, 2003; Saint-Germain and Greene, 2009; Nappi, 2011). Salvage also removes fire-killed trees that would contribute to fuel loading and future fire risk, a perennial concern near rural communities that has become more urgent in recent years (Nicholls *et al.*, 2006; Chapin *et al.*, 2008).

Widespread application of clearcutting to produce a sustained timber yield creates an unnatural age structure that retains no old growth stands on a landscape scale, as has been observed in Sweden (Esseen *et al.*, 1992). Older successional stages in the boreal forest are associated with higher levels of biodiversity (Kneeshaw and Gauthier, 2003; Burton, 2013), but sustained yield harvest in Canada's boreal forest has also reduced older age classes (Venier *et al.*, 2014). A shift from green harvest toward salvage allows uncut/unburned stands to persist past typical rotation ages (Bergeron *et al.*, 2004). When properly planned, salvage harvest of productive forests can reduce the harvest of unburned stands while simultaneously sustaining forest economies and enhancing landscape-scale stand age diversity.

Salvage harvest of burned forest stands may have clear economic benefits, but it is less clearly positive in its ecological effects (McRae *et al.*, 2001; Lindenmayer *et al.*, 2008; Nappi, 2011). Salvage constitutes a second disturbance to the burned forest, and does not emulate natural disturbance effects well enough to be a viable ecological forestry practice in itself (Schmiegelow *et al.*, 2006). Salvage typically occurs within a few years of wildfire before residual timber degrades in value due to insects, fungi, and weathering (Saint-Germain and Greene, 2009; Nappi, 2011). The process of salvage during the sensitive tree recruitment phase may alter stand composition and successional trajectory by changing key site or seedbed parameters, which negatively affects the resilience of the ecosystem (Boucher *et al.*, 2014). For example, salvage harvest may reduce diversity of post-fire specialist species, introduce invasive species, increase grass cover, and inhibit root-sprouting species by homogenizing post-fire stands through a more uniform light, moisture, and temperature environment (Kurulok and Macdonald, 2004). Road access must be created to transport timber (Saint-Germain and Greene, 2009), and heavy machinery entry to remove timber simultaneously affects soil characteristics, remnant stand structure, and legacy seed- and budbanks (Greene *et al.*, 2006; Boucher *et al.*, 2014). Harvest activity alters natural seedbed characteristics by exposing more mineral soil and increasing the available microsites for seed germination, particularly on skid trails or when the ground is unfrozen (Greene *et al.*, 2006; Martin-DeMoor *et al.*, 2010). Soil disturbance from harvest machinery and seedling burial by logging residue reduced conifer seedlings in post-fire Cascadian conifer stands (Donato *et al.*, 2006). Mechanical damage from harvest entry also eliminates early conifer germinants (Greene *et al.*, 2006), and reduces aspen suckering by 60% compared to burned-only sites (Fraser *et al.*, 2004). Other studies have found that the salvage removals themselves impact nutrient cycling (Brais *et al.*, 2000) and homogenize an

otherwise patchy post-burn mosaic of seedbed and wildlife habitat conditions (Greene *et al.*, 2006; Schmiegelow *et al.*, 2006; Nappi, 2011). Salvage harvest is increasingly common across many fire-prone ecosystems (Lindenmayer *et al.*, 2004; Nappi *et al.*, 2004), necessitating silvicultural practices that mediate some of its negative ecological consequences.

Ecological forestry expressly articulates the need to ensure biological legacies and suitable recovery periods that typify post-disturbance succession (Franklin *et al.*, 2007). By coupling salvage harvest with assisted regeneration, longer recovery periods are permitted in unharvested green stands and biological legacies may be approximated through accelerating reforestation. Assisted regeneration is defined as those silvicultural practices that "modify the structure and composition to accelerate stand recovery and succession" (Le Goff *et al.*, 2005). Assisted regeneration consists of a suite of silvicultural practices, such as site preparation and artificial stocking, which replicate typical post-fire conditions to facilitate the reproduction, recruitment, and growth of a target species or plant community.

Boreal tree species primarily evolved in fire-dependent ecosystems and establish quickly following a fire. Some species have serotinous cones that constitute an aerial seedbank (e.g., *Picea mariana* Mill., *Pinus banksiana* Lamb., and *Pinus contorta* Dougl.), while others readily re-sprout from roots and stumps (e.g., *Betula neoalaskana* Sarg. and *Populus tremuloides* Michx.) (Greene *et al.*, 1999; Saint-Germain and Greene, 2009). In addition, deciduous tree seeds germinate well on thin organic layers < 2.5 cm thick (Johnstone and Chapin, 2006), with high severity disturbance exposing favorable seedbeds and increasing seedling recruitment (Zasada, 1985; Harvey and Bergeron, 1989; Perala and Alm, 1990). Site preparation to expose mineral soil seedbeds in conjunction with an available seed source enhances recruitment by emulating natural post-fire conditions, especially for *B. neoalaskana* (Zasada and Grigal, 1978; Safford *et al.*, 1990; Wurtz and Zasada, 2001). For species that rely on asexual reproduction like *P. tremuloides*, silvicultural practices focus on stimulating sprouting by killing the aboveground portions of the tree and reducing vegetative competition, similar to fire (Perala, 1990; Shepperd, 2001). Other boreal species like *Picea glauca* Moench [Voss] may recolonize a site only by seed dispersal from nearby mature individuals, with most reproduction within 100 m of the seed source (Dobbs, 1976; Rupp, 1997; Gärtner *et al.*, 2011). Within a fire perimeter, heterogeneous burn severity, unburned islands, and irregular burn edges constitute important seed sources (Saint-Germain and Greene, 2009; Nappi, 2011). Adequate stocking of *P. glauca* is less certain as distance increases, but improves with the availability of seedbeds with thin organic layers (Zasada, 1985; Zasada *et al.*, 1992; Johnstone and Chapin, 2006). Successful regeneration practices for *P. glauca* often involve seedbed modifications to provide suitable germination microsites and suppress competing vegetation. Where a

disturbance eliminates proximate seed sources, assisted stocking by direct seeding or planted seedlings are utilized (Putman and Zasada, 1986; Lieffers *et al.*, 1996; Cole *et al.*, 1999; Densmore *et al.*, 1999). Forest managers can ameliorate low post-disturbance reforestation through the application of assisted regeneration practices that consider the unique reproductive biology of each boreal tree species.

As an adaptation strategy, assisted regeneration builds resilience into the forest structure by accomplishing the re-establishment of species whose means of reproduction may be reduced or eliminated by climate change or salvage harvest while the site remains otherwise capable of growing the species. Assisted regeneration practices are particularly important where natural regeneration may be deficient or lagging, for example the seed source is removed through harvest (Greene *et al.*, 2006) or eliminated by intense fire (Johnstone *et al.*, 2010b). Finally, intact forest cover is linked to social benefits (e.g., timber products, clean water) (Turner *et al.*, 2012), and reforestation minimizes the time until forest cover is re-established. Forest management that incorporates assisted regeneration can reduce the risk of lost social benefits from purely stochastic factors.

Ecological management of the boreal forest may be fostered by the increased use of salvage harvest and assisted regeneration to sustain ecosystem productivity and its social analogs. However, up-to-date and operationally relevant silvicultural practices are essential to address unknown future climate and disturbance regimes, uncertain probabilities of various successional pathways, and interactions among multiple factors within the ecosystem (Ogden and Innes, 2009; Chapin *et al.*, 2010). Researchers have developed many structured decision-making and adaptive management frameworks, and these stress the importance of weighing alternatives, monitoring a course of action, and continually re-evaluating management actions (Ohlson *et al.*, 2005; Ogden and Innes, 2007; Swanston *et al.*, 2012). One common theme of these adaptation frameworks is the importance of science-based knowledge about the effects of silvicultural practices over time. To obtain operationally relevant knowledge, it is critical to implement and compare different practices. For example, Yukon Territory forest practitioners advocated operational-scale trials of management alternatives (including site preparation, planting, salvage harvest) as a crucial tool to close the knowledge gap surrounding climate change adaptations (Ogden and Innes, 2009). Large-scale field experiments are a critically important type of ecosystem research, that represent manipulations of interest to managers at an operational scale and "are large enough to include the relevant physical, chemical, and biotic context of the processes being studied" (Carpenter, 1998). These operational-scale field experiments help identify alternatives and tradeoffs in the application of silvicultural practices, and provide information to enhance forest management (Carpenter, 1998; Monserud, 2002; Puettmann *et al.*, 2009).

In the following chapter, I investigate a 26.7 ha experiment that was established following a high severity wildfire and salvage harvest. I evaluated 60 unique combinations of treatments across three factor levels: topographic position, site preparation of the seedbed, and assisted stocking of *P. glauca*. I investigated how these factors interact across different timber species, and whether effects observed shortly after tree establishment persist after 28 growing seasons. In the final chapter, I discuss some of the management implications for Alaskan foresters resulting from this study.

The boreal forest is one of the largest biomes in the world, with continuous intact unmanaged ecosystems (Hare and Ritchie, 1972; Moen *et al.*, 2014). In contrast to the high-intensity management regimes of Fennoscandia (Esseen *et al.*, 1992; Angelstam, 1998) and southern Canada (Bergeron and Fenton, 2012; Bell, 2015), Interior Alaska's boreal forests have been only lightly impacted by harvesting activities (Wurtz *et al.*, 2006). Although global climate change is altering Interior Alaska's disturbance regime as well as plant community dynamics and other ecosystem processes, managers have the opportunity to undertake ecologically-sound silviculture while sustaining the productive capacity of the ecosystem for future generations. My hope is that the analysis of the operational-scale experiment evaluated in this thesis will contribute to improved boreal silvicultural practices for managing the forest resource amid a changing climate.

1.2 References

Angelstam, P.K., 1998. Maintaining and restoring biodiversity in European boreal forests by developing natural disturbance regimes. Veg. Sci. 9, 593-602. doi: 10.2307/3237275.

Barber, V.A., Juday, G.P., Finney, B.P., 2000. Reduced growth of Alaskan white spruce in the twentieth century from temperature-induced drought stress. Nature 405, 668-673. doi: 10.1038/35015049.

Beck, P.S.A., Juday, G.P., Alix, C., Barber, V.A., Winslow, S.E., Sousa, E.E., Heiser, P., Herriges, J.D., Goetz, S.J., 2011. Changes in forest productivity across Alaska consistent with biome shift. Ecol. Lett. 14, 373-379. doi: 10.1111/j.1461-0248.2011.01598.x.

Bell, F.W., 2015. The Effects of the intensification of silviculture on plant diversity in northern temperate and boreal forests of Ontario, Canada. In, Environ. Sci. University of Guelph, Guelph, ON, p. 406.

Bergeron, Y., Fenton, N.J., 2012. Boreal forests of eastern Canada revisited: old growth, nonfire disturbances, forest succession, and biodiversity. Botany 90, 509-523. doi: 10.1139/b2012-034.

Bergeron, Y., Flannigan, M., Gauthier, S., Leduc, A., Lefort, P., 2004. Past, current and future fire frequency in the Canadian boreal forest: implications for sustainable forest management. Ambio 33, 356-360. doi: 10.1579/0044-7447-33.6.356.

Bergeron, Y., Leduc, A., Harvey, B.D., Gauthier, S., 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forest. Silva Fennica 36, 81-95. doi: N/A.

Blanco, G., Eby, M., Edmonds, J., Fleurbaey, M., Gerlagh, R., Kartha, S., Kunreuther, H., Rogelj, J., Schaeffer, M., Sedláček, J., Sims, R., Ürge-Vorsatz, D., Victor, D., Yohe, G., 2014. Climate Change 2014 Synthesis Report. Intergovernmental Panel on Climate Change, United Nations

Boucher, D., Gauthier, S., Noël, J., Greene, D.F., Bergeron, Y., 2014. Salvage logging affects early post-fire tree composition in Canadian boreal forest. For. Ecol. Manage. 325, 118-127. doi: 10.1016/j.foreco.2014.04.002.

Brais, S., David, P., Ouimet, R., 2000. Impacts of wild fire severity and salvage harvesting on the nutrient balance of jack pine and black spruce boreal stands. For. Ecol. Manage. 137, 231-243. doi: 10.1016/S0378-1127(99)00331-X.

Burton, P.J., 2013. Exploring complexity in boreal forests, in: (Eds.) Messier, C., Puettmann, K.J., Coates, K.D., Managing Forests as Complex Adaptive Systems. Routledge, New York, pp. 79-109.

Calef, M., Varvak, A., McGuire, A., Chapin, F., Reinhold, K., 2015. Recent changes in annual area burned in Interior Alaska: The impact of fire management. Earth Interactions In press. doi: 10.1175/EI-D-14-0025.

Carpenter, S.R., 1998. The need for large-scale experiments to assess and predict the response of ecosystems to perturbation, in: (Eds.) Pace, M.L., Groffman, P.M., Successes, Limitations, and Frontiers in Ecosystem Science. Springer, New York, pp. 287-312.

Chapin, F.S., Callaghan, T.V., Bergeron, Y., Fukuda, M., Johnstone, J.F., Juday, G.P., Zimov, S.A., 2004. Global change and the boreal forest: thresholds, shifting states or gradual change? Ambio 33, 361-365. doi: 10.1579/0044-7447-33.6.361.

Chapin, F.S., McGuire, A.D., Ruess, R.W., Hollingsworth, T.N., Mack, M.C., Johnstone, J.F., Kasischke, E.S., Euskirchen, E.S., Jones, J.B., Jorgenson, M.T., Kielland, K., Kofinas, G.P., Turetsky, M.R., Yarie, J., Lloyd, A.H., Taylor, D.L., 2010. Resilience of Alaska's boreal forest to climatic change. Can. J. For. Res. 40, 1360-1370. doi: 10.1139/X10-074.

Chapin, F.S., Trainor, S.F., Huntington, O., Lovecraft, A.L., Zavaleta, E., Natcher, D.C., McGuire, A.D., Nelson, J.L., Ray, L., Calef, M., Fresco, N., Huntington, H., Rupp, T.S., DeWilde, L.o., Naylor, R.L., 2008. Increasing wildfire in Alaska's boreal forest: pathways to potential solutions of a wicked problem Bioscience 58, 531-540. doi: 10.1641/B58060.

Cole, E.C., Newton, M., Youngblood, A., 1999. Regenerating white spruce, paper birch, and willow in south-central Alaska. Can. J. For. Res. 29, 993-1001. doi: 10.1139/x99-030.

Densmore, R.V., Juday, G.P., Zasada, J.C., 1999. Regeneration alternatives for upland white spruce after burning and logging in interior Alaska. Can. J. For. Res. 29, 413-423. doi: 10.1139/x99-008.

Dobbs, R.C., 1976. White spruce seed dispersal in central British Columbia. Forestry Chron. 52, 225-228. doi: 10.5558/tfc52225-5.

Donato, D.C., Fontaine, J.B., Campbell, J.L., Robinson, W.D., Law, J.B.K.B.E., 2006. Post-wildfire logging hinders regeneration and increases fire risk. Science 311, 352. doi: N/A.

Drever, C.R., Peterson, G., Messier, C., Bergeron, Y., Flannigan, M., 2006. Can forest management based on natural disturbances maintain ecological resilience? Can. J. For. Res. 36, 2285-2299. doi: 10.1139/x06-132.

Esseen, P.-A., Ehnström, B., Ericson, L., Sjöberg, K., 1992. Boreal forests—the focal habitats of Fennoscandia, in: (Ed.) Hansson, L., Ecological principles of nature conservation. Elsevier, London, pp. 252-325.

Franklin, J.F., Mitchell, R.J., Palik, B.J., 2007. Natural disturbance and stand development principles for ecological forestry. U.S. Department of Agriculture, Forest Service

Fraser, E., Landhäusser, S., Lieffers, V., 2004. The effect of fire severity and salvage logging traffic on regeneration and early growth of aspen suckers in north-central Alberta. Forestry Chron. 80, 251-256. doi: 10.5558/tfc80251-2.

Gärtner, S.M., Lieffers, V.J., Macdonald, S.E., 2011. Ecology and management of natural regeneration of white spruce in the boreal forest. Environ. Rev. 19, 461-478. doi: 10.1139/a11-017.

Greene, D.F., Gauthier, S., Noël, J., Rousseau, M., Bergeron, Y., 2006. A field experiment to determine the effect of post-fire salvage on seedbeds and tree regeneration. Front. Ecol. Environ. 4, 69-74. doi: 10.1890/1540-9295(2006)004[0069:AFETDT]2.0.CO;2.

Greene, D.F., Zasada, J.C., Sirois, L., Kneeshaw, D., Morin, H., I. Charron, Simard, M.-J., 1999. A review of the regeneration dynamics of North American boreal forest tree species. Can. J. For. Res. 29, 824-839. doi: 10.1139/x98-112.

Hare, F.K., Ritchie, J.C., 1972. The boreal bioclimates. Geographical Rev. 62, 333-365. doi: 10.2307/213287.

Harvey, B.D., Bergeron, Y., 1989. Site patterns of natural regeneration following clear-cutting in northwestern Quebec. Can. J. For. Res. 19, 1458-1469. doi: 10.1139/x89-222.

Heinselman, M.L., 1981. Fire and succession in the conifer forests of northern North America, in: (Eds.) West, D.C., Shugart, H.H., Botkin, D.B., Forest Succession. Springer, New York, pp. 374-405.

Ilisson, T., Chen, H.Y.H., 2009. Response of six boreal trees species to standreplacing fire and clearcutting. Ecosystems 12, 820-829. doi: 10.1007/s10021-009-9259-z.

Johnstone, J.F., Chapin, F.S., 2006. Effects of soil burn severity on post-fire tree recruitment in boreal forest. Ecosystems 9, 14-31. doi: 10.1007/s10021-004-0042-x.

Johnstone, J.F., Chapin, F.S., Hollingsworth, T.N., Mack, M.C., Romanovsky, V., Turetsky, M., 2010a. Fire, climate change, and forest resilience in interior Alaska. Can. J. For. Res. 40, 1302-1312. doi: 10.1139/X10-061.

Johnstone, J.F., Hollingsworth, T.N., Chapin, F.S., Mack, M.C., 2010b. Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. Glob. Change Biol. 16, 1281-1295. doi: 10.1111/j.1365-2486.2009.02051.x.

Johnstone, J.F., Rupp, T.S., Olson, M., Verbyla, D., 2011. Modeling impacts of fire severity on successional trajectories and future fire behavior in Alaskan boreal forests. Landsc. Ecol. 26, 487-500. doi: 10.1007/s10980-011-9574-6.

Juday, G., Barber, V., Vaganov, E., Rupp, S., Sparrow, S., Yarie, J., Linderholm, H., 2005. Forests, land management, and agriculture, in: (Ed.) Symon, C., Arctic Climate Impact Assessment. Cambridge University Press, New York, pp. 781-862.

Juday, G.P., Barber, V., Rupp, T.S., Zasada, J.C., Wilmking, M., 2003. A 200-year perspective of climate variability and the response of white spruce in Interior Alaska, in: (Eds.) Greenland, D., Goodin, D.G., Smith, R.C., Climate variability and ecosystem response at long-term ecological research sites. Oxford University Press, New York, NY, pp. 226-250.

Kasischke, E.S., Bergen, K., Fennimore, R., Sotelo, F., Stephens, G., Jaentos, A., Shugart, H.H., 1999. Satellite imagery gives clear picture of Russia's boreal forest fires. Eos 80, 141-152. doi: 10.1029/99EO00094.

Kasischke, E.S., Rupp, T.S., Verbyla, D.L., 2006. Fire trends in the Alaskan boreal forest, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 285-301.

Kasischke, E.S., Turetsky, M.R., 2006. Recent changes in the fire regime across the North American boreal region: spatial and temporal patterns of burning across Canada and Alaska. Geophysical Res. Lett. 33, 1-5. doi: 10.1029/2006GL025677.

Kasischke, E.S., Verbyla, D.L., Rupp, T.S., A. David McGuire, Murphy, K.A., Jandt, R., Barnes, J.L., Hoy, E.E., Paul A. Duffy, Calef, M., Turetsky, M.R., 2010. Alaska's changing fire regime - implications for the vulnerability of its boreal forests. Can. J. For. Res. 40, 1313-1324. doi: 10.1139/X10-098.

Kimmins, J.P., 2004. Forest ecology: a foundation for sustainable forest management and environmental ethics. Macmillan Publishing Company, Upper Saddle River, NJ.

Kneeshaw, D., Gauthier, S., 2003. Old growth in the boreal forest: a dynamic perspective at the stand and landscape level. Environ. Rev. 11, 99-114. doi: 10.1139/a03-010.

Kneeshaw, D.D., Leduc, A., Drapeau, P., Gauthier, S., Pare, D., Carignan, R., Doucet, R., Bouthillier, L., Messier, C., 2000. Development of integrated ecological standards of sustainable forest management at an operational scale. Forestry Chron. 76, 481-493. doi: 10.5558/tfc76481-3.

Kolb, T.E., Wagner, M.R., Covington, W.W., 1994. Concepts of forest health. Forestry 92, 10-15. doi: N/A.

Kurulok, S., Macdonald, E., 2004. Impacts of post-fire salvage logging on tree regeneration and plant communities in the mixedwood boreal forest of Alberta Sustainable Forest Management Network

Le Goff, H., Leduc, A., Bergeron, Y., Flannigan, M., 2005. The adaptive capacity of forest management to changing fire regimes in the boreal forest of Quebec. Forestry Chron. 81, 582-592. doi: 10.5558/tfc81582-4.

Lemmen, D.S., Warren, F.J., 2004. Climate change impacts and adaptation: a Canadian perspective. Climate Change Impacts and Adaptation Directorate, Natural Resources Canada

Lieffers, V.J., Stewart, J.D., Macmillan, R.B., Macpherson, D., Branter, K., 1996. Semi-natural and intensive silvicultural systems for the boreal mixedwood forest. Forestry Chron. 72, 286-292. doi: 10.5558/tfc72286-3.

Lindenmayer, D.B., Burton, P.J., Franklin, J.F., 2008. Salvage Logging and its Ecological Consequences. Island Press, Washington, DC.

Lindenmayer, D.B., Foster, D.R., Franklin, J.F., Hunter, M.L., Noss, R.F., Schmiegelow, F.A., Perry, D., 2004. Salvage harvesting policies after natural disturbance. Science 303, 1303. doi: N/A.

Martin-DeMoor, J., Lieffers, V.J., Macdonald, S.E., 2010. Natural regeneration of white spruce in aspen-dominated boreal mixedwoods following harvesting. Can. J. For. Res. 40, 585-594. doi: 10.1139/X10-016.

McRae, D.J., Duchesne, L.C., Freedman, B., Lynham, T.J., Woodley, S., 2001. Comparisons between wildfire and forest harvesting and their implications in forest management. Environ. Rev. 9, 223-260. doi: 10.1139/er-9-4-223.

Millar, C.I., Stephenson, N.L., Stephens, S.L., 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecol. Applications 17, 2145-2151. doi: 10.1890/06-1715.1.

Moen, J., Rist, L., Bishop, K., Chapin, F.S., Ellison, D., Kuuluvainen, T., Petersson, H., Puettmann, K.J., Rayner, J., Warkentin, I.G., Bradshaw, C.J.A., 2014. Eye on the taiga: removing global policy impediments to safeguard the boreal forest. Conservation Lett. 7, 408-418. doi: 10.1111/conl.12098.

Monserud, R.A., 2002. Large-scale management experiments in the moist maritime forests of the Pacific Northwest. Landsc. Urban Plan. 59, 159-180. doi: 10.1016/S0169-2046(02)00013-0.

Nappi, A., 2011. Harvesting in burned forests — issues and orientations for ecosystem-based management. Ministère des Ressources naturelles et de la Faune, Direction de l'environnement et de la protection des forêts

Nappi, A., Drapeau, P., Savard, J.-P.L., 2004. Salvage logging after wildfire in the boreal forest: Is it becoming a hot issue for wildlife? Forestry Chron. 80, 67-74. doi: 10.5558/tfc80067-1.

Nicholls, D.L., Patterson, S.E., Uloth, E., 2006. Wood and coal cofiring in Interior Alaska: utilizing woody biomass from wildland defensible-space fire treatments and other sources. U.S. Department of Agriculture, Forest Service PNW-RN-551.

Ogden, A.E., Innes, J., 2007. Incorporating climate change adaptation considerations into forest management planning in the boreal forest. Int. Forestry Rev. 9, 713-733. doi: 10.1505/ifor.9.3.713.

Ogden, A.E., Innes, J.L., 2009. Adapting to climate change in the southwest Yukon: locally identified research and monitoring needs to support decision-making on sustainable forest management. Arctic Institute of North America 62, 159-174. doi: 10.2307/40513285.

Ohlson, D.W., McKinnon, G.A., Hirsch, K.G., 2005. A structured decision-making approach to climate change adaptation in the forest sector Forestry Chron. 81, 97-103. doi: 10.5558/tfc81097-1.

Perala, D.A., 1990. Populus tremuloides Michx. quaking aspen, in: (Eds.) Burns, R.M., Honkala, B.H., Silvics of North America: Volume 2, Hardwoods. U.S. Department of Agriculture, Forest Service, Washington DC, p. 34.

Perala, D.A., Alm, A.A., 1990. Regeneration silviculture of birch: a review. For. Ecol. Manage. 32, 39-77. doi: 10.1016/0378-1127(90)90105-K.

Pickett, S.T.A., White, P.S., 1985. Natural disturbance and patch dynamics: an introduction, in: (Eds.) Pickett, S.T.A., White, P.S., The ecology of natural disturbance and patch dynamics. Academic Press, Orlando, FL, pp. 3-16.

Puettmann, K.J., Coates, K.D., Messier, C., 2009. A Critique of Silviculture. Island Press, Washington DC.

Putman, W.E., Zasada, J.C., 1986. Direct seeding techniques to regenerate white spruce in interior Alaska. Can. J. For. Res. 16, 660-664. doi: 10.1139/x86-115.

Rupp, T.S., 1997. A geographic model of landscape-level post-disturbance forest establishment patterns of Interior Alaska white spruce ecosystems. In, School of Natural Resources. University of Alaska, Fairbanks.

Safford, L.O., Bjorkbom, J.C., Zasada, J.C., 1990. Betula papyrifera Marsh. paper birch, in: (Eds.) Burns, R.M., Honkala, B.H., Silvics of North America: Volume 2, Hardwoods. U.S. Department of Agriculture, Forest Service, Washington DC, p. 19.

Saint-Germain, M., Greene, D.F., 2009. Salvage logging in the boreal and cordilleran forests of Canada: Integrating industrial and ecological concerns in management plans. Forestry Chron. 85, 120-134. doi: 10.5558/tfc85120-1.

Schmiegelow, F.A., Stepnisky, D.P., Stambaugh, C.A., Koivula, M., 2006. Reconciling salvage logging of boreal forests with a natural-disturbance management model. Conservation Biol. 20, 971-983. doi: 10.1111/j.1523-1739.2006.00496.x.

Seymour, R.S., Hunter, M.L., Jr., 1999. Principles of ecological forestry, in: (Ed.) Hunter, M.L., Jr., Maintaining biodiversity in forested ecosystems. Cambridge University Press, Cambridge UK, pp. 22-61.

Shepperd, W.D., 2001. Manipulations to regenerate aspen ecosystems. In: Shepperd, W.D., Binkley, D., Bartos, D.L., Stohlgren, T.J., Eskew, L.G. (Eds.), Sustaining Aspen in Western Landscapes: Symposium Proceedings. Rocky Mountain Research Station, U.S. Department of Agriculture, Forest Service, Grand Junction, CO, pp. 355-366.

Soja, A.J., Tchebakova, N.M., French, N.H.F., Flannigan, M.D., Shugart, H.H., Stocks, B.J., Sukhinin, A.I., Parfenova, E.I., Chapin, F.S., Stackhouse Jr, P.W., 2007. Climate-induced boreal forest change: Predictions versus current observations. Glob. Planet. Change 56, 274-296. doi: 10.1016/j.gloplacha.2006.07.028.

Spittlehouse, D.L., Stewart, R.B., 2003. Adaptation to climate change in forest management. BC J. Ecosyst. Manage. 4, 1-11. doi: N/A.

Swanston, C., Janowiak, M., Butler, P., Parker, L., St. Pierre, M., Brandt, L., 2012. Forest adaptation resources: climate change tools and approaches for land managers. U.S. Department of Agriculture, Forest Service GTR-NRS-87.

Turner, M.G., Donato, D.C., Romme, W.H., 2012. Consequences of spatial heterogeneity for ecosystem services in changing forest landscapes: priorities for future research. Landsc. Ecol. 28, 1081-1097. doi: 10.1007/s10980-012-9741-4.

Venier, L.A., Thompson, I.D., Fleming, R., Malcolm, J., Aubin, I., Trofymow, J.A., Langor, D., R. Sturrock, Patry, C., Outerbridge, R.O., Holmes, S.B., Haeussler, S., Grandpré, L.D., Chen, H.Y.H., E. Bayne, Arsenault, A., Brandt, J.P., 2014. Effects of natural resource development on the terrestrial biodiversity of Canadian boreal forests. Environ. Rev. 22, 457-490. doi: dx.doi.org/10.1139/er-2013-0075.

Weber, M.G., Flannigan, M.D., 1997. Canadian boreal forest ecosystem structure and function in a changing climate: impact on fire regimes. Environ. Rev. 5, 145-166. doi: 10.1139/a97-008.

Wurtz, T.L., Ott, R.A., Maisch, J.C., 2006. Timber harvest in Interior Alaska, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 302-308.

Wurtz, T.L., Zasada, J.C., 2001. An alternative to clear-cutting in the boreal forest of Alaska: a 27-year study of regeneration after shelterwood harvesting. Can. J. For. Res. 31, 999-1011. doi: 10.1139/cjfr-31-6-999.

Zasada, J.C., 1985. Production, dispersal, and germination of white spruce and paper birch and first year seedling establishment after the Rosie Creek Fire, in: (Eds.) Juday, G.P., Dyrness, C.T., Early Results of the Rosie Creek Fire Research Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK, pp. 34-37.

Zasada, J.C., Grigal, D., 1978. The effects of silvicutural system and seedbed preparation on natural regeneration of white spruce and associated species. In, Proceedings of the Fifth North American Forest Biology Workshop, 1978, Gainesville, Fla. Southern Forest Experiment Station, U.S. Department of Agriculture, Forest Service, Gainesville, FL, pp. 213-220.

Zasada, J.C., Sharik, T.L., Nygren, M., 1992. The reproductive process in boreal forest trees, in: (Eds.) Shugart, H., Leemins, R., Bonan, G., A systems analysis of the global boreal forest. Cambridge University Press, Cambridge, UK, pp. 85-125.

Chapter 2 – Experimental assisted regeneration treatments in the stem exclusion stage of a burned and salvaged Interior Alaska boreal forest[1]

2.1. Abstract

Wildfire and subsequent timber salvage harvests are forecasted to increase in the Alaska boreal forest, creating the need to evaluate the effectiveness of forest regeneration practices in light of these interacting disturbances. Silvicultural practices such as site preparation and assisted regeneration are carried out shortly after disturbance in order to ensure satisfactory reforestation. Thirty years after a stand-replacing wildfire and salvage, we sampled an operational-scale (26.7 ha) white spruce (*Picea glauca* Moench [Voss]) regeneration trial established in a productive upland forest. Regeneration treatments were applied in a split-split plot experimental design on two landform types (LF), four ground scarification treatments (GST) plus a non-scarified control, and five artificial white spruce regeneration treatments (WSRT) plus a natural seedfall control. We analyzed total biomass as well as stand density and basal area for all species, seeking to evaluate the persistence of regeneration silvicultural treatment effects 28 growing seasons after installation. Our results indicate GST had no significant effect on white spruce basal area or stem density. However, compared to natural seedfall control plots, white spruce basal area was six times higher in planted seedling plots, and white spruce stem density (dbh ≥ 1.0 cm) was nearly three times greater in broadcast seeding plots. White spruce stem density from natural seedfall averaged 944 stems ha^{-1}, but density was dependent on both topographic position and distance to wind-dispersed seed sources. Scarification nearly doubled Alaska birch (*Betula neoalaskana* Sarg.) stem density and basal area compared to non-scarified control plots. Planted white spruce plots supported 19% less birch basal area, except in the most intensive scarification treatments in which birch basal area did not differ. Although quaking aspen (*Populus tremuloides* Michx.) density and basal area were generally unaffected by the regeneration silvicultural practices in this study, intensive scarification reduced basal area by half on slope plots. These results confirm that regeneration silvicultural practices continue to influence stand development beyond the stem initiation stage, but practices that promote one species may reduce others.

[1] Prepared for submission in Forest Ecology and Management as: Allaby, A., Juday, G., Yarie, J., and Young, B. Experimental assisted regeneration treatments in the stem exclusion stage of a burned and salvaged Interior Alaska boreal forest.

2.2. Introduction

Silvicultural practices, which target tree recruitment as well as subsequent forest composition and growth, are the principal tools used by forest managers to meet human objectives (Puettmann *et al.*, 2009). Effective silvicultural practices, which produce lasting effects on stand composition, are especially critical in boreal ecosystems where low productivity and thin economic margins place great importance on efficient, low-intensity management (Wurtz *et al.*, 2006). Boreal forest succession after disturbance is often characterized by re-establishment of the same plant community through a relay floristic pathway (Viereck *et al.*, 1986; Chapin *et al.*, 2006b; Kurkowski *et al.*, 2008). Tree regeneration of all species typically occurs during a short post-disturbance window due to accumulating organic layers and rapid canopy closure (Galipeau *et al.*, 1997; Johnstone *et al.*, 2004; Johnstone and Chapin, 2006), but species dominance shifts as the ecosystem moves through seral stages (Chapin *et al.*, 2006b). Practices carried out during the early stem recruitment stage are thought to exert considerable influence species composition and eventual harvest volume (Hawkins *et al.*, 2006; Cortini *et al.*, 2010). In Interior Alaska, a changing fire regime (Kasischke *et al.*, 2010; Beck *et al.*, 2011) and increasing wood energy use (Fresco and Chapin, 2009; DOF, 2010; AEA, 2011) are placing additional demands on forest resources that may necessitate greater management intensity.

Stand-replacing wildfire is the principal disturbance to the boreal forest (Heinselman, 1981), and boreal trees species have developed reproductive strategies suited to fire (Greene *et al.*, 1999). Relatively stable fire regimes over the past 6,000 years (Kasischke *et al.*, 2006) sorted boreal tree species into community assemblages adapted to particular fire frequencies, sizes, and severities (Heinselman, 1981; Weber and Flannigan, 1997). However, a rapidly changing climate is shifting the fire regime toward greater area burned, higher severity, and shorter return intervals across not only Alaska but all of North America's boreal forest driven by a warmer climate (Flannigan *et al.*, 2005; Kasischke and Turetsky, 2006; Balshi *et al.*, 2009; Kasischke *et al.*, 2010). Fire regime changes render less certain the typical successional processes of post-disturbance stands by altering the reproductive environment, particularly through uncharacteristic combinations of forest floor characteristics and propagule availability (Chapin *et al.*, 2004; Chapin *et al.*, 2010; Johnstone *et al.*, 2010; Johnstone *et al.*, 2011).

The practice of salvage harvesting burned areas in Alaska will likely increase with an accelerating fire regime, as has been documented in the intensively managed Canadian boreal forest (Lindenmayer *et al.*, 2004; Nappi *et al.*, 2004). Already salvage harvest is an important practice for Alaska's timber industry (Crimp *et al.*, 1997; DOF, 2013; Hermanns, 2013). From 1973-2012, 16% of all Interior Alaska harvested acreage was post-fire salvage, with more total salvage acreage in years where large fires

burned accessible productive forests (Morimoto *et al.*, unpublished data). Salvage harvest, however, constitutes a second disturbance to the ecosystem and presents a distinct regeneration environment (Lindenmayer *et al.*, 2008; Boucher *et al.*, 2014). Heavy equipment entry and stem removal that accompany salvage harvest result in soil compaction, increased rates of erosion, disturbed organic layers, elimination of post-fire tree regeneration, more light and evaporation at the forest floor, and altered nutrient cycling (Saint-Germain and Greene, 2009; Nappi, 2011). Wildfire typically results in a patchy seedbed environment with variable organic layer thickness (Zasada *et al.*, 1983; Greene *et al.*, 2007). By contrast, salvage harvest typically removes residual structure and homogenizes the seedbed environment (Kurulok and Macdonald, 2004; Greene *et al.*, 2006). Silvicultural research examining post-fire or post-harvest sites alone does not adequately reflect the interacting disturbance effects of post-fire salvage harvest.

Assisted regeneration practices suited to Alaska's boreal forest have been studied only since the 1970s (see Zasada, 1976; Wurtz *et al.*, 2006; Juday *et al.*, 2013). Much of this previous research focused on the stand initiation stage of forest development for a single tree species, white spruce (*Picea glauca* Moench [Voss]) (see for example Zasada and Grigal, 1978; Zasada and Packee, 1995; Wurtz *et al.*, 2006). Successful seedling recruitment during this early stage depends on the presence of sufficient propagules on site and, in the case of sexual reproduction, the availability of suitable seedbeds for germination (Zasada, 1986; Oliver and Larson, 1996). However, the stem exclusion stage, during which trees grow into saplings and the canopy closes, involves intense competition between individuals and species for light, moisture, and nutrients (Zasada and Packee, 1995; Chen and Popadiouk, 2002). Characteristics measured early in a stand's development may not account for future stand conditions within boreal mixed forests for three reasons. First, early regeneration treatments may attenuate over time. For example, in British Columbia spruce planted on scarified and non-scarified seedbeds demonstrated similar total volume at 15 years despite early differences (Bedford *et al.*, 2000). Also, height advantages of planted white spruce seedlings in this experiment observed at 5 years were no longer detectable at 20 years (Boateng *et al.*, 2006). Second, unintended results of treatments may become apparent later in stand development. For example, scarification preceding an unusually large seed crop resulted in overstocked stands with stunted trees after 27 years (Wurtz and Zasada, 2001). Third, some boreal stand types experience extended recruitment periods of moderately shade-tolerant conifers. For example, quaking aspen (*Populus tremuloides* Michx.) canopies let in sufficient light to permit continued white spruce establishment, typically reaching maximum stocking more than 20 years post-fire (Youngblood, 1995; Lieffers *et al.*, 1996). Significant recruitment may occur on decomposing logs,

further lengthening the recruitment window (Lieffers *et al.*, 1996; Peters *et al.*, 2006). Forest regeneration research to identify effective silvicultural practices for Alaska would benefit from examining all tree species in the mixed boreal forest, and at later stages of forest development.

The objective of this study is to evaluate the effectiveness of assisted regeneration practices after 28 years which were employed in Interior Alaska's productive upland forest type (Viereck *et al.*, 1983). To assess this objective, we analyze an assisted regeneration trial that was established following a stand-replacing wildfire which occurred in 1983 and was then subsequently salvage harvested (Densmore *et al.*, 1999). The regeneration trial was conceived to examine "the survival, distribution, and growth of white spruce seedlings…among five site preparation methods and six regeneration methods" (Densmore *et al.*, 1999). This study was conducted at the Rosie Creek Fire Tree Regeneration Installation (RCFTRI), the largest operational comparison of silvicultural practices in boreal Alaska known to the authors. Its well-replicated and controlled experimental design allows us to compare the outcomes of treatments applied shortly after disturbance. The post-fire salvage sites we examined present a relatively homogeneous environment to test differing practices while minimizing confounding factors. Throughout this study, we explicitly test the null hypothesis of no treatment effect, thus evaluating whether early treatments have an enduring effect on stand structure and composition. Utilizing the parent experiment permits comprehensive evaluation of all factorial combinations of silvicultural treatments for each of the major upland timber species –aspen, Alaska birch (*Betula neoalaskana* Sarg.), and white spruce – in terms of both volume and stem density. In view of the shifting demand from raw material for wood products to energy, we also examine the biomass implications of different practices. Finally, extending analysis to all upland timber species clarifies the impacts of silvicultural practices on the future of burned and salvaged forests.

2.3. Methods

2.3.1.Study Area

The boreal forest of Interior Alaska has only a few tree species, but the distribution of these species and the community associations they form is spatially complex. Boreal species and community distributions are strongly impacted by slope, aspect, elevation, and disturbance history (Van Cleve *et al.*, 1991; Chapin *et al.*, 2006a; Kurkowski *et al.*, 2008). South-facing uplands support some of the most productive forest ecosystems, particularly white spruce, mixed white spruce-hardwood, and hardwood stands (Van Cleve *et al.*, 1983; Viereck *et al.*, 1983; Viereck *et al.*, 1986). These forest types comprise 63% of state-managed forestry lands in Interior Alaska (Crimp *et al.*, 1997), but only 15% of Interior

Alaska forest (Hammond, 1996). Upland forests are the focus of much harvest activity, in which the principal commercial tree species are white spruce, Alaskan birch , and quaking aspen (Wurtz et al., 2006).

The study area consists of the Rosie Creek Fire Tree Regeneration Installation (RCFTRI), a 26.7 ha experiment established in 1985-6 to examine white spruce seedling recruitment under a variety of stand initiation practices within two distinct upland topographic types (Densmore *et al.*, 1999). RCFTRI is located 30 km southwest of Fairbanks, Alaska at 148.31°W 64.74°N, in the Bonanza Creek Experimental Forest (Figure 2.1). The area is comprised of the "interior forested lowlands and uplands" ecoregion of the boreal forest (Gallant *et al.*, 1995). Rolling hills and valleys marked by a cold continental climate, discontinuous permafrost, and high wildfire frequency form a mosaic of ecosystems (Beget *et al.*, 2006) that encompass some of Interior Alaska's most productive forestland (Wurtz *et al.*, 2006).

The study area extends across mostly south-facing uplands above the Tanana River, and has deep, permafrost-free, silt-loam soils of aeolian origin (Soil Survey Staff, 2011). The site experiences an average July temperature of 16.1°C, an average January temperature of -19.0°C, and annual rainfall of 200 mm (Van Cleve *et al.*, 2013a, b). An estimated 35% of annual precipitation falls as snow over the winter (Viereck and Slaughter, 1986). RCFTRI experienced stand-replacing high severity crown fire on 2 June 1983 as part of the greater 3,482 ha Rosie Creek wildfire (Juday, 1985). Subsequent clearcut salvage logging operations took place on the entire installation during the snow-free season ending in August 1985 (Densmore *et al.*, 1999).

2.3.2.Experimental Design

The RCFTRI was designed as a blocked split-split plot experiment, comprising four hierarchical nested levels (Figure 2.2) (Densmore *et al.*, 1999). The first level involved two upland landform types (LF), ridge and slope, corresponding to the whole plot factor (approximately 13.3 ha each). Each LF whole plot was partitioned into three blocks (approximately 4.4 ha each). Each block was further divided into five equal split plots (approximately 0.9 ha each) where one of five ground scarification treatments (GST) were randomly assigned. Each GST split plot was subsequently further divided into six equal split-split plots (S/SP; approximately 0.15 ha each, or 40 x 40 m) where one of six white spruce regeneration treatments (WSRT) were randomly assigned (Densmore *et al.*, 1999). There were three replicates (n=3), each represented by a split-split plot (S/SP), for every unique combination of the three factor levels.

Though only one kilometer apart, the ridge and slope LF have distinct topography and spatial configuration that may hold important silvicultural implications. Ridgetop split-split plots (S/SPs) range

between 311-344 m elevation, while slope S/SPs extend from 194-291 m. On the ridge, 59 of 90 S/SPs have a slope angle < 5°, compared to only 10 of 90 slope S/SPs. Within both LFs, S/SPs have similar spatial configurations to the unburned forest edge, but many ridge S/SPs are much further from wind-dispersed seed sources (Table 2.1). Prevailing wind direction during autumn seed dispersal was quantified for a nearby floodplain site with equal frequency from northeast and southwest (Youngblood and Max, 1992).

The ground scarification treatments (GST) involved mechanical disturbance of the residual organic layer of the forest floor and exposure of mineral soil seedbeds. Five GSTs were established in each block in August/September 1985: one non-scarified control and four types of mechanical scarification. Control S/SPs were not scarified, and mineral soil exposure was estimated at 0% in 1985 (Densmore *et al.*, 1999). Organized from most to least mineral soil exposed, the four scarification types included: 1) bulldozer blading in parallel strips (29% mineral soil exposed), 2) double-disc trenching with a TTS-35 Disc Trencher in perpendicular passes (25%), 3) single-disc trenching in parallel passes (15%), and 4) patch scarification with a Bracke-type scarifier (10%). The mineral soil exposed within each of the GSTs did not significantly differ between the ridge and slope, nor did the percent of vegetation cover, treatment depth, or the spatial configuration of the treatments (for details see Densmore *et al.*, 1999).

The white spruce regeneration treatments (WSRT, refer to Figure 2.2) introduced white spruce propagules collected in 1983 from nearby unburned stands. Each of the six different WSRTs were carried out during 1986 on every split plot, and these included: natural seedfall (control), planted seedlings, broadcast seeding, spring unsheltered spot seed, fall unsheltered spot seed, and spring sheltered spot seed. The control WSRT was stocked by natural white spruce seed dispersal, which is highly episodic (Greene *et al.*, 1999). Two-year-old containerized seedlings, which came from seeds collected from mature trees near the planting sites and seeded in the Alaska State Forest Nursery in Palmer, Alaska, were planted at 2.4-m spacing (8 feet, or one tree every 6 m^2) in June 1986. The planted seedling stock in Ridge Blocks 2 and 3 was stunted due to contamination with a common greenhouse fungus (Densmore *et al.*, 1999). Broadcast seeding occurred at a rate of 1 kg ha^{-1}. Spring and fall spot seedings involved three to eight seeds placed in favorable microsites at the same spacing as planted seedlings. Sheltered seed spots were placed under photodegradable plastic cones that lasted approximately one year in order to exclude seed predators. Spot seeding methods and seed shelters were a common regeneration practice employed in Scandinavian forests (Putman and Zasada, 1986), and their use in this experiment represented a large-scale operational test in boreal Alaska.

2.3.3.Data Collection

We sampled a portion of each experimental unit's surface area to estimate forest composition. Sampling took place from July to September 2013, with supplemental sampling in early June 2014, and comprised two overlapping datasets. The first dataset, referred to as 3WSRT, consists of the three most commonly practiced WSRTs – natural seedfall, broadcast seeding, and planted seedlings – balanced across all six blocks (90 S/SPs). The second dataset, referred to as 6WSRT, includes the three common WSRTs plus the three experimental spot seeding WSRTs from three of the six blocks (90 S/SPs total; Ridge Block 1, Ridge Block 3, and Slope Block 1). These two datasets share 45 S/SPs in common – the S/SPs that represent the three commonly practiced WSRTs in Ridge Block 1, Ridge Block 3, and Slope Block 1 – summing to a total 135 sampled S/SPs of 180 in the parent experiment.

Each S/SP was systematically sampled by establishing a two-by-two crosscutting pattern of four 1-m wide belt transects on each S/SP which accounted for the heterogeneous forest composition within each plot (discussed below, 2.4.1). Key site characteristics may demonstrate gradients or a patchy distribution even at small scales (Reed *et al.*, 1993), even within a single 40 m x 40 m S/SP in an even-aged forest. Heterogeneity in observed tree distribution and growth may be attributable to variables such as the light environment (Deutschman *et al.*, 1999), burn severity (Johnstone and Chapin, 2006), soil nutrient characteristics (Lechowicz and Bell, 1991), legacy effects such as sprouting from birch or aspen root stock (Greene *et al.*, 1999), or differential vegetative competition within patches (Cater and Chapin, 2000). The orientation of some ground scarification treatments in parallel strips, as well as the presence of topographical gradients may further contribute to within-plot variability.

Transect layout (Figure 2.3) was established by 1) identifying the plot boundary on an azimuth of between 0 and 90°, and dividing it into thirds, 2) along the selected boundary at the one-third and two-thirds positions, transects one and two were placed perpendicular to the selected plot boundary, 3) the northernmost transect was then divided into thirds, and transects three and four were placed perpendicular to transects one and two. On average, we sampled 10.1% of a S/SP's total area with this method.

Within the 1-m belt transects, we recorded species and diameter-at-breast-height (dbh = 1.37 m) for every live tree or shrub where 1) ≥ 50% of root collar was within the belt transect, and 2) dbh ≥ 1.0 cm. Shrubs including willows (*Salix spp.*) and alders (*Alnus spp.*) were identified to genus. We selected the minimum diameter both for efficiency and to sample principally those trees which may form the current or future canopy. Measurement protocols followed those previously established within the region (Malone *et al.*, 2009).

To validate our transect sampling method, we also conducted a complete census on six S/SPs by recording species and dbh for every live tree and shrub ≥ 1.0 cm dbh within the plot boundaries, according to the inclusion and measuring protocols above. One S/SP was censused in each block, representing each of the five GSTs and four of the six WSRTs. Direct comparison of census and transect estimates permits assessment of any systematic bias in the sampling method.

Comparisons between experimental units of basal area or biomass calculated from dbh-only equations require both a good correlation between height and dbh (Crow and Schlaegel, 1988), and that the functional relationship between the two remains the same regardless of experimental treatment. For example, discrete height-diameter relationships have been observed across stands with varying species composition, basal area, or stem density (Huang and Titus, 1994; Sharma and Zhang, 2004), all characteristics that may be impacted by our experimental factors. To assess the equivalence of height-diameter relationships within each species, we measured tree height in addition to dbh on 14 S/SPs. We selected a balance of plots across the ridge and slope LF, scarified and non-scarified GSTs, and the three most commonly practiced WSRTs (i.e., broadcast-seeding, planted seedling, and natural seedfall control). On each S/SP we performed a systematic sample of every 20th tree within the plot boundary, recording height, dbh, and species on all live stems ≥ 1.0 cm dbh. Small amounts of willow, balsam poplar, and black spruce were present but excluded from the systematic sample because of their low proportions, and alder shrubs were also omitted because the biomass equations only utilize diameter.

Due to the effects of topography on tree growth and stand composition (Viereck *et al.*, 1986; Burnett *et al.*, 1998; Kurkowski *et al.*, 2008), we derived elevation, slope, and aspect from a 5-m pixel size digital elevation model (Alaska, 2010) in ArcMap 10.1 (ESRI, 2012). At high latitudes such as this study, forest community types are significantly influenced by slope aspect, with east- and west-facing slopes hosting different communities than south-facing aspects (Kurkowski *et al.*, 2008). However, this experiment had almost exclusively east- or south-facing aspects, with only four north-facing S/SPs and six west-facing (and those in one block only). Since the experiment does not include a balanced representation of aspects, we incorporated the influence of aspect by deriving cumulative growing season solar radiation. Solar radiation aggregates the effects of latitude, slope angle, and slope aspect into a single figure: the amount of usable energy in watt-hours m^{-2} (Hinzman *et al.*, 2006). We calculated cumulative annual solar radiation in ArcMap 10.1 using the 'Area Solar Radiation' tool for the period 1 April – 30 September (calculated at 14-day intervals, using 0.5 hour interval during sampled days, and with 30% cloud cover) (ESRI, 2012).

Seed dispersal distance is also an important factor controlling sexual regeneration of boreal tree species, and total dispersal distance is a function of prevailing wind direction (Zasada, 1985; Greene *et al.*, 1999). In order to include this variable in our dataset, we classified a 22 August 1986 color infrared image into burned and unburned areas (USGS, 1986), then partitioned unburned areas into northeast and southwest seed sources relative to each LF whole plot (Youngblood and Max, 1992). Wind-dispersed seed sources were further classified as containing hardwoods only, or mixed spruce/hardwood communities (AHAP, 1988). We calculated several measures of seed dispersal distance to S/SP centroid: 1) distance to nearest unburned seed source, 2) the mean distance from southwest and northeast seed sources, 3) minimum distance to closest wind-dispersed seed source, and 4) minimum distance to closest spruce seed source.

2.3.4.Statistical Analysis

To assess the validity of our transect method, we used paired t-tests to evaluate the presence of mean bias in the sample transect estimates compared to the census results, similar to methods employed by Huang *et al.* (2000). Given no systematic bias in our transect method, we then evaluated the effects of the experimental factors in three ways. First, the appropriate ANCOVA (Analysis of Covariance) model was constructed and the effects of the three experimental factors – LF, GST, WSRT – as well as the two- and three-way interactions were tested. Second, post hoc differences between treatment means were evaluated with Dunnett's test, which compares treatments to a control. Third, we selected a set of relevant treatment contrasts (e.g., scarified vs. non-scarified) and tested whether the assembled groups demonstrated a significant difference in the response variables (i.e., stem density). We carried out two linear contrasts for each response variable, and used the Bonferroni method to control Type 1 error rates (significance level: p=0.05/k for k contrasts) (Kuehl, 2000).

For analyzing the 3WSRT dataset, we used a mixed effects ANCOVA model with balanced data, employing a blocked split-split plot design with block was treated as a random effect (Kuehl, 2000). The experimental unit was the split-split plot (S/SP), and each S/SP was treated as equal and comparable even though S/SPs differed somewhat in geometry from the idealized 40 x 40 m square. Statistical analysis was performed within SAS software, Version 9.3, using Proc GLM (SAS, 2012). All hypothesis-testing used a Type I error rate $\alpha = 0.05$ for significance, though p-values between 0.05 and 0.10 were labeled marginally significant and merit consideration.

For analyzing the 6WSRT dataset, we used a mixed ANCOVA model with balanced data, employing a blocked split plot design with block as a random effect. Analysis paralleled the 3WSRT

dataset, but the model was reduced by omitting the LF (whole plot) factor, as there were insufficient degrees of freedom to test hypotheses about LF effect.

The ANCOVA model is an ANOVA with the addition of split-split plot covariates to reduce experimental error and increase precision of hypothesis tests. Inclusion of each covariate into the final model was assessed sequentially in three ways: 1) the covariate is independent of the experimental treatments, 2) a simple linear regression between the response variable and the covariate indicates a significant linear relationship, and 3) the covariate term achieves significance in the overall ANCOVA (Kuehl, 2000). We tested eight spatial covariates using the balanced 3WSRT dataset: 1) elevation, 2) slope angle, 3) cumulative solar radiation, 4) distance to unburned seed source, 5) natural logarithm of distance to unburned seed source, 6) distance to nearest wind-dispersed seed source, 7) mean distance to wind-dispersed seed sources, and 8) natural logarithm of spruce distance to spruce seed source. We used log-transformed variables to test seed dispersal relationships found by Densmore *et al.* (1999).

All S/SP response variables were calculated on a per hectare basis, and values are reported as mean ± 1 SE unless noted otherwise. All response variables were power-transformed using the Box-Cox method to meet parametric assumptions. Constant variance was evaluated on model residuals with the Brown-Forsyth Test of treatment medians. We analyzed eight response variables: stem density (stems ha^{-1}) and basal area (m^2 ha^{-1}) individually for aspen, birch, and white spruce; total stem density for all species; and total aboveground woody plant biomass (kg ha^{-1}) for all species. We calculated aboveground biomass dry weight using national allometric equations from Jenkins *et al.* (2003) for all species except alder. Alder biomass was derived from locally calibrated equations (Wurtz, 1995). We are confident that the national biomass equations reasonably estimate biomass of Interior Alaska trees, as equations from Jenkins *et al.* (2003) track very closely to those calibrated locally by Yarie *et al.* (2007); however, the localized equations were not used because they did not include the minimum diameters used in our study.

We tested the height-diameter relationship in a multiple regression considering broad categories of experimental factors. Similar the analysis of full and reduced ANOVAs by Pillsbury (1995) to examine regionally calibrated height-diameter models, we compared the estimated slope coefficient from a reduced model, in which height is a function of dbh alone (Equation 2.1), to the slope coefficient from a full model, in which height is a function of dbh and a categorical indicator variable from among the experimental factors (Equation 2.2):

$$\text{height} = \beta_0 + \beta_1 * \text{dbh} \quad (2.1)$$

$$\text{height} = \beta_0 + \beta_1 * \text{dbh} + \beta_2 * x_i + \beta_3 * \text{dbh} * x_i \quad (2.2)$$

where β_0 = intercept coefficient, β_1 = slope coefficient, β_2 = interaction between indicator variable and intercept, β_3 = interaction between indicator variable and slope, and x_i = indicator variable for category i. Indicator variables were created for three categories: 1) ridge vs. slope LFs, 2) scarified vs. non-scarified GST, and 3) comparisons among planted seedling, broadcast seeding, and natural seedfall WSRTs. If the β_3 term was significant, we would conclude the slope of the linear relationship between dbh and height depended on the experimental factor, represented by the indicator variable. Shade-tolerant white spruce experience relatively low mortality from hardwood canopy closure (Comeau *et al.*, 2003), whereas light competition steadily eliminates overtopped hardwood trees intolerant of shade (LaBonte and Nash, 1978; Comeau *et al.*, 2005). Therefore, in order to conform to parametric assumptions, both height and dbh were log-transformed for white spruce, while only dbh was log-transformed for aspen and birch (Crow and Schlaegel, 1988).

2.4. Results

2.4.1.Census and Transect Comparisons

No significant difference was detected between the transect versus the census data in terms of stem density and basal area for all of the different tree species ($p > 0.05$) (Table 2.2). We are confident the transect sampling accurately characterizes forest composition in each S/SP. However, alder were slightly over-represented by the belt-transect method. Five of the six census S/SPs contained fewer stems ha^{-1} than the transect estimates, and differences between the two methods were marginally significant ($p=0.082$ for alder stem density, and $p=0.095$ for alder basal area). Alders accounted for 0.8% of estimated total biomass across the 135 measured S/SPs.

2.4.2.Height Diameter Relationships for White Spruce, Birch, and Aspen

Height data, along with dbh, were collected for 230 white spruce, 340 birch, and 113 aspen on the 14 S/SPs that comprise the height dataset (Table 2.3). Using these measurements, we determined that diameter explained a large proportion of height variation: model R^2 of height- diameter regression was over 0.92 for white spruce, 0.88 for birch, and 0.77 for aspen. We found no evidence to support different slope coefficients for the height-diameter relationship for aspen and white spruce across any tested factor, or for birch across WSRT (Table 2.4). However, we found slope of the height-diameter relationship in birch was significantly different between scarified vs non-scarified and ridge vs. slope. Birch on scarified S/SPs were slightly taller at a given diameter than those growing on non-scarified S/SPs, and birch on the ridge were slightly taller than those growing on the slope. At the mean birch

diameter (dbh = 54.7 mm), a tree growing on the ridge was 9.6% taller (0.82 m) than on the slope, and 8.4% taller (0.73 m) on scarified S/SPs.

2.4.3.Spatial Covariates

2.4.3.1. Effects of Topographical Covariates on Experimental Results

No spatial covariates satisfied all ANCOVA criteria, and therefore were not included in the overall ANOVA model. However, several covariates, particularly for seed dispersal measures and solar radiation, demonstrated a significant linear relationship between the response variable and the covariate (Table 2.5), and provide additional context for our results.

2.4.3.2. Effects of Solar Radiation on Experimental Results

Though not significant for white spruce, cumulative growing season solar radiation significantly explained about 5% of the variation for both birch and aspen basal area. However, birch basal area was inversely correlated and aspen was positively correlated with solar radiation. A simple linear regression comparing birch and aspen biomass excluding planted seedling S/SPs indicates a significant negative relationship ($F_{1,103}$=113.81, p<0.0001), with 52% of the variation in aspen biomass explained by birch biomass (Figure 2.5b).

The ridge and slope LFs had similar cumulative growing season solar radiation, with 624,622 ± 2,117 watt-hours m^{-2} on the ridge compared to 625,475 ± 4,282 on the slope. Two opposite trends tended to equalize the solar radiation budgets of the two LFs. The ridge had a lower slope angle (4.8° ± 2.0°) than the slope site (8.9° ± 3.4°), but the ridge was directly south-facing while the slope had a southeast aspect (Figure 2.1). The slope site contained the 13 S/SPs with the highest incoming solar radiation values. Overall the S/SPs on the ridge experienced a narrower range of solar radiation than the slope site.

2.4.3.3. Effects of Distance to Seed Source on Experimental Results

Log distance to nearest wind-dispersed spruce seed source was the best predictor, in terms of R^2, for white spruce stem density on the 30 natural seedfall S/SPs (Table 2.5). The 30 natural seedfall S/SPs averaged 944 white spruce stems ha^{-1}, but the 12 S/SPs within 200 m of a wind-dispersed white spruce seed source averaged 1,512. For example, the seven natural seedfall S/SPs with no recorded white spruce seedlings averaged only 111 m from the unburned forest edge, well within typical spruce dispersal distances (Greene *et al.*, 1999), but were located an average of 649 m from wind-dispersed

spruce seed sources (Figure 2.1). Current Alaska reforestation standards of 1,111 stems ha^{-1} (DOF, 2007) could be met on this site for white spruce alone out to 265 m downwind from a spruce seed source according to the regression relationship. In the 3WSRT dataset, the strongest predictor of birch stem density as measured by R^2 was log distance to the unburned forest (Table 2.5, Figure 2.1). The 79% of S/SPs within 200 m of the unburned edge supported an average of 6,958 stems ha^{-1}, compared to the 21% of S/SPs that were further than 200 m and supported only 1,586 birch stems ha^{-1}. While for aspen, there was no significant relationship found for any of the distance to seed source measures.

2.4.4.Biomass and Stem Density of All Species Combined

The LF factor influenced total biomass, where values were on average less on the slope than on the ridge (72,146 ± 2,782 kg ha^{-1} (n=45) on the slope compared to 81,582 ± 2,422 (n=45) on the ridge). Though the 13% average difference is not inconsequential, the LF effect was only marginally significant (p=0.0737, Table 2.6). This trend would have to be tested in another study due to lack of replication of the LF whole plots in this design. However, total biomass by LF can be clarified by the contribution of birch and aspen, which have opposite responses to landform. Compared to the slope, birch had 102% more biomass on the ridge while aspen had 60% less biomass.

Total aboveground biomass was influenced predominantly by the GST factor in the 3WSRT dataset (i.e., natural seedfall control, broadcast seeding, and planted seedlings across all 6 blocks) (Table 2.6). Biomass was significantly higher (30%) on scarified S/SPs compared to non-scarified S/SPs (Table 2.7). Total biomass from every type of scarification treatment was significantly greater than the non-scarified control with one exception (Figure 2.4a). Biomass on blade-scarified S/SPs was not significantly greater than the non-scarified control (Figure 2.4a), but still mean biomass of blade-scarified S/SP was 23% greater than non-scarified S/SPs. Total biomass by GST can be clarified by the contribution of birch and aspen, which have opposite responses to scarification. Compared to non-scarified S/SPs, birch had 70% more biomass on scarified S/SPs while aspen had 14% less biomass.

WSRT did not significantly affect total biomass in the overall ANOVA (Table 2.6). Though planting shade-tolerant spruce could establish fuller site occupancy and increase total biomass compared to the other natural seedfall or broadcast WSRTs, we did not detect a significant effect on total biomass (Table 2.7). Though WSRT did not increase total biomass, neither did it decrease biomass. A simple linear regression of white spruce to hardwood tree biomass in S/SPs receiving the planted seedling WSRT demonstrates a significant negative relationship (Figure 2.5). The inverse relationship indicates a direct

tradeoff between hardwood and white spruce biomass, and white spruce biomass explains 61% of the variation in hardwood biomass on planted seedling S/SPs.

The LF factor had no significant influence on total stem density, but the GST factor had marginally significant effects (p=0.0719, Table 2.6). Scarified S/SPs (combination of all treatments) had significantly greater stem density (51%) than non-scarified S/SPs (Table 2.7). Blade-scarified S/SPs in particular supported significantly greater stem density (78%) compared to non-scarified S/SPs (Figure 2.4a).

Total stem density was significantly influenced by the WSRT factor (Table 2.6). Total stem density on broadcast seed S/SPs was significantly greater (9,943 ± 673 stems ha^{-1}, n=30) compared to the combination of planted seedling and natural seedfall S/SPs (8,574 ± 540, n=60) (Table 2.7). The higher total stem density (all species) on broadcast S/SPs compared to natural seedfall treatments appears to be accounted for largely by the 1,234 stem ha^{-1} increase in white spruce stems in broadcast seeded S/SPs.

2.4.5.Shrub and Other Tree Species Biomass and Stem Density

The woody shrubs measured in this study, alders and willows, comprised a small portion of total biomass and stem density. The willows, in particular, appeared to be preferentially browsed by moose and were of poor form and low stature. Averaged across the 3WSRT dataset, willows comprised 1.4% of stems and 0.1% of biomass. Alders comprised 8.9% of stems and 0.9% of biomass. None of the experimental treatment factors had a significant effect on biomass or stem density for either shrub. In addition, we recorded 2 black spruce (*Picea mariana* Mill.) and 12 balsam poplar (*Populus balsamifera* L.) among the 18,100 trees that were included in the 135 sampled S/SPs, a negligible fraction of total stem density and basal area. These species were included in the total biomass figures for all species combined, but did not receive additional study.

2.4.6.White Spruce Basal Area and Stem Density

The LF factor by itself did not significantly influence either white spruce basal area or stem density, but LF had a significant interaction with WSRT (Table 2.6). In general, there was greater magnitude of white spruce basal area and stem density on slope S/SPs compared to ridge S/SPs, although the rank-ordering of WSRTs within both LF types was the same (Figure 2.7b). Stem densities of planted seedling S/SPs on both ridge and slope were comparable, but white spruce stem density in absolute terms was more than three times greater on the slope for both broadcast seeding and natural

seedfall S/SPs. However, on both the ridge and slope, the broadcast seed WSRT resulted in proportionally similar increases in stem density compared to natural seedfall (152% increase on the ridge, 159% increase on the slope).

The GST factor alone had did not have a significant impact on white spruce stem density in the overall ANOVA (Table 2.6, Figure 2.4b), nor did the linear contrast of scarified vs. non-scarified GSTs (Table 2.7). However, broadcast seeding following any type of scarification treatment produced significantly higher stem densities than non-scarified natural seedfall controls (Figure 2.8b). Broadcast seeding of non-scarified S/SPs resulted in statistically similar white spruce stem densities compared to the natural seedfall treatment (Figure 2.8b). The positive response of white spruce stem density to the combination of broadcast seeding and scarification was apparent on the slope. Broadcast seeded, scarified slope S/SPs averaged 4,355 ±655 stems ha^{-1} (n=12) compared to non-scarified S/SPs with 1,727 ± 449 (n=3). The same trend was not visible on the ridge site, where broadcast seeded, scarified S/SPs averaged 925 ± 284 (n=12) compared to non-scarified S/SPs with 1,465 ± 740 (n=3). However, the three-way interaction term between LF, GST, and WSRT was not significant (Table 2.6). Scarification was associated with reduced white spruce stem density on natural seedfall S/SPs, though not significantly (Figure 2.8b). One type of scarification, double-disc, appeared to reduce white spruce basal area on non-planted S/SPs. Broadcast seeded S/SPs supported less white spruce basal area (29%) on double-disc scarified sites compared to the non-scarified controls, and natural seedfall S/SPs receiving the double-disc treatment supported 54% less basal area compared to the non-scarified controls, although the effect was not significant in either case (Figure 2.8b).

White spruce basal area within the 3WSRT dataset was significantly influenced by both WSRT and LF after 28 growing seasons (Table 2.6). Broadcast seeding and planted seedling treatments significantly increased basal area relative to the natural seedfall control (Dunnett, $p<0.0001$). Planted seedling S/SPs supported an average of 7.44 ± 0.93 m^2 ha^{-1} (n=30) white spruce basal area, broadcast seeding resulted in 2.67 ± 0.52 (n=30), while natural seedfall had the lowest basal area at 1.25 ± 0.25 (n=30) (Figure 2.6a). The planted seedling S/SPs located in the two blocks impacted by greenhouse fungus at the time of planting (Densmore *et al.*, 1999) supported only 38% of the white spruce basal area compared to planted seedling S/SPs in the four unaffected blocks.

WSRT had a highly significant effect on white spruce stem density (Table 2.6), with both planted seedling and broadcast WSRTs producing significantly greater white spruce stem densities compared to natural seedfall (Dunnett, $p<0.0001$). Broadcast seeding resulted in 2,431 ± 412 stems ha^{-1} (n=30),

compared to planted seedling S/SPs with 1,601 ± 161 (n=30) and natural seedfall with 944 ± 187(n=30) (Figure 2.6b).

The 6WSRT dataset includes only three blocks, within which a balanced set of S/SPs were sampled for all six white spruce regeneration treatments. Analysis of the 6WSRT dataset permits inferences about the three experimental spot seeding methods, which have not been presented in results thus far based on the 3WSRT dataset. The only significant experimental factor that influenced white spruce basal area within the 6WSRT dataset, based on the overall ANOVA, was WSRT (Table 2.8). Even though the 6WSRT dataset includes one of the two ridgetop blocks that received fungus-infected planted seedlings, planted seedling S/SPs still supported high white spruce basal area (6.26 ± 1.01 m^2 ha^{-1}, n=15). White spruce basal area on planted seedling S/SPs was significantly higher than the natural seedfall S/SPs, which averaged only 1.00 ± 0.31 (n=15) (Dunnett, $p<0.0001$). Basal area on the three spot seed WSRTs and broadcast seeding treatment combined (1.40 ± 0.24, n=60) were similar to natural seedfall (Dunnett, $p>0.1742$) (Figure 2.6a). However, on Slope Block 1 both broadcast seeding and sheltered spot seeding treatments supported higher white spruce basal area, 85% and 174% respectively, compared to the natural seedfall S/SPs (Figure 2.6a). On Ridge Blocks 1 and 3, the basal area increase in broadcast seeding (59%) and sheltered spot seeding (16%) treatments compared to natural seedfall was lower than on Slope Block 1. However, we did not have sufficient replicates across both LFs for all six WSRT to estimate an interaction term.

As with basal area, the only significant experimental factor for estimating white spruce stem density within the 6WSRT dataset in the overall ANOVA was WSRT (Table 2.8). Broadcast seeding (Dunnett, $p=0.0111$) and planted seedlings (Dunnett, $p=0.0074$) had significantly higher stem densities than natural seedfall (Figure 2.6b). Spruce stem density in the S/SPs that received the two (spring and fall) unsheltered spot seed treatments was indistinguishable from natural seedfall (Dunnett, $p>0.05$). Sheltered spot seeding produced comparable white spruce stem density to the planted seedling treatment, but was not significantly different from the natural seedfall control (Dunnett, $p=0.0922$). The LF factor may affect spruce stem density, but the 6WSRT dataset does not allow a direct test. Slope S/SPs receiving the sheltered spot seeding treatment supported much greater spruce stem density (3,821 ± 955 stems ha^{-1}, n=5) compared to ridge S/SPs (727 ± 156, n=10). Overall, the substantial differences for shelter spot seeding observed between ridge and slope S/SPs generated high variability for this WSRT, with nearly twice the standard deviation (1934.1 stems ha^{-1}) of the planted seedling treatment (1096.4 stems ha^{-1}). Combining all WSRTs, slope S/SPs supported more white spruce stems (2,561 ± 306, n=30) compared to ridge S/SPs (807 ± 91, n=60).

2.4.7.Birch Basal Area and Stem Density

All three levels of experimental factors influenced birch basal area and stem density within the 3WSRT dataset (Table 2.6). The LF effect on basal area was consistent (no interaction) regardless of GST and WSRT. Slope S/SPs supported less birch basal area (7.82 ± 1.12 m^2 ha^{-1}, n=45) than ridge S/SPs (16.48 ± 0.81, n=45). The WSRT factor significantly influenced birch basal area in the overall ANOVA (Table 2.6). The planted seedling WSRT resulted in lower birch basal area overall (10.74 ± 1.33, n=30) compared to broadcast seeding (12.93 ± 1.48, n=30) and natural seedfall (12.78 ± 1.49, n=30) treatments (Figure 2.7c). Birch basal area was significantly lower (19%) on S/SPs receiving the planted seedling WSRT compared to the combination of broadcast and natural seedfall (Table 2.7).

Although the GST factor alone did not significantly influence birch basal area, the effect of GST was mediated by a significant interaction with WSRT (Table 2.6). Scarified S/SPs had significantly greater (75%) birch basal area than non-scarified S/SPs (Table 2.6). More intensive scarification was associated with greater birch basal area, particularly the double-disc S/SPs which supported significantly greater (105%) birch basal area compared to non-scarified controls (Figure 2.4c). Blade scarification supported an average of 91% greater basal area compared to non-scarified controls, but the difference was not significant compared to non-scarified controls (Figure 2.4c).

The presence of a significant interaction between GST and WSRT interaction indicated that scarification treatments produced different effects based on the WSRT (Table 2.6). In particular, birch basal area on planted seedling S/SPs depended on the intensity of scarification (Figure 2.8c). Non-scarified planted seedling S/SPs supported less than half the birch basal area of scarified planted seedling S/SPs (Table 2.7). On the planted seedling S/SPs, only the double-disc treatment produced significantly greater birch basal area than the non-scarified, natural seedfall controls (Figure 2.8c). In addition, on planted seedling S/SPs, the intensive GSTs (i.e., blade and double-disc) were associated with greater birch basal area (13.86 ± 2.06 m^2 ha^{-1}, n=36) than on moderate GSTs (i.e., patch and single-disc; 10.19 ± 2.05, n=18). Although intensive scarification produced the greatest birch basal area on planted seedling S/SPs, white spruce did not show any concomitant reduction in basal area (Table 2.7).

Birch stem density tended to be higher on the ridge LF (8,208 ± 3,587 stems ha^{-1}, n=45) than on the slope (3,439 ± 3,507, n=45), though the LF factor was not significant in the overall ANOVA (Table 2.6). Although GST was not significant in the overall ANOVA (Table 2.6), the contrast of all scarification treatments combined compared to the non-scarified treatment was significant; scarification of any kind increased birch stem density by 98% (Table 2.7). Intensive GSTs were associated with significantly more

birch stems than the non-scarified control (Figure 2.4c), but the less intensive scarification methods of patch and single-disc were not significantly different from the non-scarified control. WSRT was not a significant factor influencing birch stem density in the overall ANOVA (Table 2.6). Birch stem density in planted seedling S/SPs was only slightly less than natural seedfall S/SPs, and overall birch stem density was not significantly influenced by the planted seedling WSRT (Table 2.7).

2.4.8.Aspen Basal Area, Biomass, and Stem Density

Aspen distribution was extremely variable across the study area. On the ridge LF, no aspen were recorded in the sample transects for 21 of 45 S/SPs, and on the slope 15 of 45 lacked aspen. The data for both aspen stem density and basal area were highly right skewed, with 20 of 90 S/SPs exceeding mean stem density (5 of which were on the ridge) and 22 of 90 S/SPs exceeding mean basal area (6 of which were on the ridge).

The experimental factors had limited impacts on aspen basal area and density within the 3WSRT dataset. The LF factor was a marginally significant predictor of aspen basal area (p=0.0901) (Table 2.6). The ridge LF supported less aspen basal area (2.37 ± 0.76 m^2 ha^{-1}, n=45) than the slope (6.31 ± 1.18, n=45).

The GST factor did not significantly influence aspen basal area (Table 2.6), though some trends merit reporting. Intensive GST (i.e., blade and double-disc scarification) tended to reduce aspen basal area by 40% on average compared to non-scarified controls, though these effects were not significant possibly due to limitations of the dataset (Table 2.7).

The effect of GST on aspen basal area was mediated by a marginally significant interaction with LF (p=0.0875, Table 2.6). Intensive scarified S/SPs on the slope supported less than half the aspen basal area (3.61 ± 1.68 m^2 ha^{-1}, n=18) compared to non-scarified S/SPs (8.00 ± 2.28, n=9). Intensive scarification on planted seedling S/SPs was associated with 77% less aspen basal area compared to non-scarified, natural seedfall controls, though the effect was not significant (Figure 2.8d).

Aspen basal area on natural seedfall S/SPs (5.41 ± 1.51m^2 ha^{-1}, n=30) tended to be nearly twice the level of planted seedling S/SPs (3.03 ± 0.93, n=30). However, the WSRT factor did not exert significant influence over aspen basal area (Table 2.6). Also, when compared directly, aspen basal area on planted seedling S/SPs was not significantly different from the natural seedfall control (Dunnett, p>0.05).

An ANOVA of aspen biomass clarifies some of the tendencies suggested by the aspen basal area results due to slightly smaller standard errors. The interaction between LF and GST was significant (Table

2.6), indicating that the response of aspen biomass to a given scarification treatment depended on which landform was being treated. The reduction of aspen biomass due to intensive scarification was significant (42%) compared to non-scarified controls (Table 2.7).

Aspen stem density was affected by similar experimental factors as basal area. Stem density was higher on slope S/SPs (1,055 ± 230 stems ha^{-1}, n=45) than on ridge S/SPs (303 ± 129, n=45), and the difference was marginally significant (p=0.0681, Table 2.6), again partially due to the patchy aspen representation within the dataset. GST alone was not a significant predictor of aspen stem density, but a significant interaction term indicated GST effects depended on LF type (Table 2.6). Intensive GST decreased aspen stem density significantly on both LFs together (Table 2.7). Slope S/SPs receiving intensive scarification had approximately half the aspen stem density (665 ± 345, n=18) of non-scarified S/SPs (1,204 ± 378, n=9). The effect appeared to be reversed on the ridge, with intensively scarified S/SPs (209 ± 75, n=18) compared to non-scarified S/SPs (90 ± 27, n=9). Planted seedling S/SPs supported 31% fewer aspen stems on average (555 ± 211, n=30) compared to natural seedfall S/SPs (810 ± 258, n=30), though the WSRT factor was not significant in the overall ANOVA (Table 2.6).

2.5. Discussion

2.5.1.Sampling Methods and Covariates

The belt transect sampling method provided an unbiased means to minimize sampling error while maintaining field efficiency. Although alder density and basal area were slightly overrepresented in transect estimates, these differences were marginally significant. Many alder ramets (stems) originate from a single individual genet, and genets demonstrate a patchy spatial distribution in Alaska's boreal forest (Wurtz, 1995). Apparently, where alders occurred within a belt transect, multiple stems were likely to be encountered, inflating estimates of their abundance. Alders accounted for a negligible amount of estimated total biomass, so even a systematic upward bias would have a minimal effect on total biomass conclusions.

Diameter-at-breast-height was a good predictor of total height for all species (Table 2.4). In this study, only birch demonstrated a significantly different slope parameter for the relationship between height and diameter, with taller birch on the ridge LF and on scarified surfaces. In Alberta increasing stand densities of white spruce and aspen were correlated with distinct height-diameter functions, primarily as a response to competition for light (Huang and Titus, 1994). Consistent with this explanation, ridge S/SPs and scarified surfaces had significantly greater birch densities in this study. Despite this detectable difference, allometric equations relating diameter and height across distinct

geographic areas are generally acceptable for differences of less than 12% (Pillsbury, 1995), and our results were well within that tolerance.

Even though each response variable was significantly correlated with some spatial covariates in a simple linear regression, none of the S/SP covariates were significant in the ANCOVA model. The original experimental design of LF whole plots and blocks accounted for geographical variables efficiently, indicating that within-whole plot and within-block variances of geographical covariates were much smaller than those over the entire experiment. RCFTRI was designed to explicitly test specific assisted regeneration silvicultural treatments, and the original researchers minimized the possible effect of exogenous factors through careful temporal control of treatment implementation, spatially-sensitive blocking, and precise survey techniques to delineate treatment boundaries (Juday *et al.*, 2013).

The inherited experimental design had some limitations for our analysis. There was only one replicate of a slope LF and one of a ridge, so hypothesis testing about LF was limited to 1 degree of freedom. However, it was important to include this variable because blocks in the same whole plot have a correlated error variance. Otherwise, blocks in the same whole plot (LF) would be treated as independent replicates, constituting a case of pseudo-replication resulting in unjustifiably narrower confidence intervals (Hurlbert, 1984). Further, it should be noted that the parent experiment was not explicitly designed to test a balanced, continuous distribution of topographic gradients (e.g., elevation, slope angle, and distance from seed sources). The original experimental design also did not explicitly control for legacy effects, such as root sprouting from aspen clones and widely variable proximity to wind-dispersed seed sources, which obscured whole plot (LF) inferences and hypothesis testing for aspen. Finally, the experiment was not tended after establishment, so that natural reproduction and successional processes complicated the effects of treatments on recruitment. Still, a lack of tending activities, such as pre-commercial thinning or ongoing vegetation management, typifies forest practices in Alaska's boreal forest (Morimoto et al., unpublished data).

2.5.2.Biomass

GST exerted major influence over total aboveground biomass after 28 growing seasons, with 30% more biomass on scarified S/SPs (Figure 2.4a). Three of the GSTs (blade scarification excluded) increased biomass significantly. Scarification reduces ground cover vegetation, which permits warmer soils and reduces competition for small tree seedlings (Zasada and Grigal, 1978; Bella, 1986). Trees in the non-blade scarification treatments apparently accumulated woody biomass more quickly than on the non-scarified control, an effect that now persisted into the stem exclusion stage. Biomass in blade

scarification treatments, however, was not significantly different from controls, largely because of a 63% reduction in aspen and a simultaneous 82% increase in birch biomass compared to non-scarified controls. We conclude that all types of scarification continued to influence stand biomass, and that blade scarification caused two opposite responses that were still measurable after nearly 30 years.

WSRT had a neutral effect on total biomass, though two countervailing trends were observed: planted seedlings increased white spruce biomass by a factor of 7 but decreased birch biomass by 23% compared to natural seedfall S/SPs (Table 2.7) and decreased aspen biomass by 46%. We interpret this result to indicate that planting seedlings shifts biomass production from hardwood seedlings to white spruce, but does not increase aggregate biomass due to the significant inverse relationship between white spruce and hardwood biomass (Figure 2.5a). Although broadcast seeding of white spruce increased total stem density compared to natural seedfall, there was no measurable effect on total biomass. Because white spruce is more shade-tolerant than either aspen or birch, it might be hypothesized that total site biomass accumulation would be augmented by an enhanced spruce component because of niche complementarity, in which increased species diversity leads to more complete site occupancy (Tilman *et al.*, 2001). However, in our study, the expected increase in biomass from fuller site occupancy apparently was small, as we found WSRT to have negligible effects on total biomass at this stage of development. Factors that have been demonstrated to be associated with higher site productivity in boreal ecosystems include soil nutrient availability, soil warmth, and moisture (Yarie and Van Cleve, 2006). Artificial introduction of white spruce propagules, whether seeding or planting, apparently did not significantly change any of these key parameters that determine site productivity.

Biomass of all species combined was marginally greater on the ridge LF than the slope (Table 2.6). The relatively hot, dry climate regime prevailing in Interior Alaska since the 1970s has been implicated in decreased spruce growth attributed to moisture deficits (Juday *et al.*, 2003; Barber *et al.*, 2004). One explanation for the productivity difference between LFs is that ridge may be buffered against high rates evaporation associated with decreased tree growth. Alternatively, the different aggregate biomass on the two LFs could have been an artifact of the relative balance of site suitability for birch (greater on the ridge) versus aspen (greater on the slope), and their divergent responses to ground scarification treatments. The great majority (80%) of S/SPs received some kind of scarification treatment. Birch biomass responded positively to scarification, so on the ridge where non-scarified controls support more birch biomass than aspen (79% vs. 11%, respectively), scarification would have promoted more biomass overall. Aspen biomass responded negatively to scarification at the highest

intensity, so on the slope where non-scarified controls support relatively less birch and more aspen (44% vs. 30%, respectively), scarification would not have elevated biomass as much.

2.5.3.White Spruce Basal Area and Stem Density

2.5.3.1. Effects of Ground Scarification Treatments

Initial (12-year) results indicated that scarification tended to increase seedling survival for all WSRT though not significantly (Densmore *et al.*, 1999). In our data, scarification appeared to exercise no influence over white spruce recruitment (as measured by stem density) from both natural seedfall and planted seedling treatments (Figure 2.8b). However, our results demonstrate that on sites treated by scarification of any type, broadcast seeding in particular did significantly increase white spruce stem density (Figure 2.8b). We interpret these results to indicate that scarification results in a substantial increase in stem density following a large (and/or even) distribution of seed after canopy removal. It is also possible that some initial white spruce seedling recruitment did not persist in scarified, natural seedfall S/SPs 16 years after the initial analysis. The ground scarification treatments employed in this study had no significant effect on white spruce basal area, maintaining the results found after 12 growing seasons (Densmore *et al.*, 1999). White spruce increases in height more slowly than birch or aspen, so competing vegetation must be suppressed for a longer time period to increase its growth significantly over hardwood competitors (Cole *et al.*, 2003). On higher productivity sites, mechanical scarification seldom suppresses competing vegetation sufficiently to allow spruce growth rates to increase (Cole *et al.*, 1999; Thiffault *et al.*, 2003; Boateng *et al.*, 2009; Wiensczyk *et al.*, 2011).

Scarification of any kind increased stem recruitment from broadcast seeding, but it did not provide a lasting vegetative control that allowed spruce to accumulate significantly more basal area. Our entire study area experienced severe wildfire and summer salvage logging, and both of these conditions increase microsites per unit area suitable for white spruce germination (Zasada, 1986; Johnstone and Chapin, 2006; Greene *et al.*, 2007; Martin-DeMoor *et al.*, 2010). Similarly, mechanical scarification treatments have been shown to increase white spruce recruitment in the first decade following canopy removal (Zasada and Grigal, 1978; DeLong *et al.*, 1997; Johnstone and Chapin, 2006). However, achieving increased growth of white spruce usually requires suppression of hardwood competitors over a more extended period than the initial scarification provides (Thiffault *et al.*, 2003; Wiensczyk *et al.*, 2011). Initial blade scarification across Interior and south-central Alaska has been shown to increase planted seedling growth for some locations, but percentage of mineral soil exposed in those treatments was double that of the most intensive (blade) treatment in this experiment (Youngblood *et al.*, 2011),

suggesting that the control of competition may have lasted longer than in our study. The importance of suppressing initial competing vegetation for an extended period is underscored by herbicide application studies, in which the effective control of competition facilitates planted seedlings to maintain measurably higher volume even after 20-30 years compared to controls (Sutton, 1995; Boateng *et al.*, 2006; Pitt *et al.*, 2010). Finally, scarification may substantially increase white spruce growth where organic layers are particularly thick (Thiffault *et al.*, 2003) or where particularly intensive mounding and capping techniques are used (Boateng *et al.*, 2006), neither of which were the case in this study.

The more intensive scarification treatments applied in our study may have stimulated hardwood competition with white spruce. Double-disc scarification tended to decrease white spruce basal area compared to no scarification, by 29% on broadcast seeding S/SPs and 54% on natural seedfall S/SPs (Figure 2.8b). However, on planted seedling S/SPs white spruce basal area was not appreciably different in controls compared to the double disc treatment, suggesting that even this intensive form of disturbance does not stimulate competition to a level that reduces planted white spruce growth. Correspondingly, double-disc scarification increased birch basal area significantly (Figure 2.4c). We believe that the early establishment of a vigorous birch canopy in these treatments suppressed white spruce growth from seed. Unfavorable light environments for understory conifer growth in mixed conifer-broadleaf stands are typically associated with slow growth and delayed canopy entry (MacIsaac and Navratil, 1995). Light transmittance through birch canopy is inversely related to birch basal area (Comeau *et al.*, 1998), and reduced light transmittance is associated with reduced white spruce basal area (Comeau *et al.*, 2003). Stands with a substantial regenerating birch component, especially stands in which the initial accumulation of birch basal area was accelerated by early scarification treatments, will almost certainly support lower levels of white spruce basal area (originated from seed) due to the early birch canopy dominance. Early establishment of planted white spruce seedlings appears to be an effective way to avoid early birch competition in burned and salvaged stands. The relative dominance of birch versus white spruce in young stands as mediated by scarification merits additional research, especially given the considerable expense of scarification.

2.5.3.2. Effects of White Spruce Regeneration Treatments

The continued growth and initiation of dominance of planted white spruce seedlings compared to all other WSRTs was one of the clearest results of the 2013-2014 re-measurement (Figure 2.6a). This effect was clear even allowing for the stunted growth due to fungal infection of spruce planting stock in two of the six blocks. We found that planted seedling S/SPs accumulated substantial basal area

regardless of GST or LF type (Figure 2.7b), an expected consequence of the elevated height and diameter growth compared to direct-seeded spruce observed after 10 years (Densmore *et al.*, 1999). Other studies have established that the success of planting white spruce seedlings on unburned sites depends on a number of factors, especially the type and amount of ground cover (Eis, 1981; DeLong *et al.*, 1997; Cater and Chapin, 2000), which can be managed through site preparation such as herbicide or mechanical scarification (Youngblood and Zasada, 1991; Cole *et al.*, 2003; Boateng *et al.*, 2009; Youngblood *et al.*, 2011). Our study establishes that planted white spruce seedlings consistently succeeded in establishing a substantial basal area advantage on burned and salvaged sites, regardless of ground cover type or scarification method.

Broadcast seeding, similar to the planted seedling WSRT, succeeded in establishing white spruce on the study area (Figure 2.6b). However, we observed many small tree seedlings on broadcast S/SPs that did not meet the measurement inclusion criteria (dbh ≥ 1.0 cm), and combined with shorter mean height and high stem density, hypothesize a substantial lag for these seedlings to reach the canopy. Our results are similar to a nearby upland experiment, in which scarification following a mast-seeding year resulted in a high input of viable seed in conjunction with suitable seedbeds, ultimately resulting in severely overstocked stands (Wurtz and Zasada, 2001). The broadcast seeding WSRT appears prone to establish overly dense stands on fire-prepared seedbeds, resulting in a crowded, slow-growing white spruce understory. Compared to broadcast seeding, planting offers greater control to channel site productivity into fewer preferred stems and accelerate a stand's basal area accumulation of white spruce (rather than non-crop species).

The slope LF appears more hospitable to white spruce recruitment from seed than the ridge, with threefold greater stem density on natural seedfall and broadcast seeded S/SPs (Figure 2.7b). Initial (5-year) white spruce survival rates for the direct seed treatments were slightly greater on the slope than the ridge, and the difference increased on the subset of S/SPs measured after 10-12 years (Densmore *et al.*, 1999). One reason for differential spruce survival by LF appears to have been the greater cover and persistence of the grass *Calamagrostis canadensis* across the ridge. Two years after treatment, disc scarification was not as effective at reducing vegetation on the ridge compared to the slope (Densmore *et al.*, 1999). *C. canadensis* grows vigorously after disturbance, cools the soil, and may suppress spruce seedling recruitment and growth (Putman and Zasada, 1986; Hogg and Lieffers, 1991; Lieffers *et al.*, 1993; Collins and Schwartz, 1998; Cater and Chapin, 2000). Conversely, the slope had greater herb cover, which is positively correlated with spruce survival and growth (Cater and Chapin, 2000). The differences in topography between the two LFs are consistent with environmental factors

that facilitate the different ground cover vegetation types. Greater solar radiation per unit area is received on steeper slopes (Hinzman *et al.*, 2006), and orographic effects induce slightly greater summer precipitation and cooler temperatures at higher elevation ridge tops (Haugen *et al.*, 1982; Viereck and Slaughter, 1986). Regardless of the ultimate cause of vegetation cover differences between the two LFs, these different vegetation covers are associated with micro-scale differences in soil temperature and light environment (Cater and Chapin, 2000; Purdy *et al.*, 2002), and we infer that these factors reduced white spruce establishment from seed on the ridge. Hypothesized boreal forest succession processes, in which slow-growing conifers eventually replace same-aged hardwood associates (Awada *et al.*, 2004; Chapin *et al.*, 2006a; Kurkowski *et al.*, 2008), appear unlikely to occur on the ridge due to the very low levels of spruce recruitment (Figure 2.6b).

Spot seeding under biodegradable shelters appears to be an effective stocking method on certain types of sites. On the slope, S/SPs receiving the seed shelter WSRT supported greater basal area than broadcast seeding despite similar stem density, and much greater basal area and stem density than natural seedfall S/SPs (Figure 2.6). On the ridge, however, sheltered spot seeding was indistinguishable from the natural seedfall. We measured one slope block and two ridge blocks for all six WSRTs, and so the number of replicates required to test the range of hypotheses involving spot seeding and LFs was insufficient. On the slope LF, initial (5-year) survival rates in sheltered seed spots were greater than in unsheltered seed spots (Densmore *et al.*, 1999). RCFTRI researchers observed seed predators consuming seed placed under seed shelters, and theorized the shelters may act as a visual cue to granivores and reduce the treatment's effectiveness (Stiell, 1976; Juday *et al.*, 2013). Seed shelters in Interior Alaska have produced highly variable results due to wildlife interactions, intrusion of moss and fungus, and flooding. However, on sites free of such challenges shelters appear to increase spruce survival (Putman and Zasada, 1986). In Ontario, spot seeding was recommended where vegetative cover was minimal and soils were light (Stiell, 1976), which better matches site characteristics of the slope whole plot compared to the ridge. The initial elevated survival of spruce in seed shelter S/SPs on the slope, detected five years after the experiment began, has now persisted into the stem exclusion stage and translated into modestly increased basal area (Figure 2.6a).

2.5.3.3. Natural Regeneration Considerations

The least-cost method to recruit spruce stems is from natural seedfall, but regeneration from natural seedfall requires both a significant seed source input and suitable seedbeds for germination (Alden, 1985; Rupp, 1997). The study area had high levels of both factors favoring white spruce

establishment: two historically large mast-seeding episodes immediately after the fire provided a strong seed source (in 1983 and 1987; see Zasada, 1985; Roland *et al.*, 2014), and highly favorable seedbed conditions compared to nearby unburned and harvested sites (Densmore *et al.*, 1999). The study area even received white spruce seed from well positioned fire perimeter tree that were fatally injured by but did not experience crown damage, and thus were able to complete the seed-masting cycle (Zasada, 1985). Finally, we found substantial white spruce recruitment throughout RCFTRI at distances up to 265 m downwind from unburned stands with a mature spruce component (see 2.4.3.3), in line with local prevailing winds (Youngblood and Max, 1992).

The orientation of the LF whole plots to the prevailing wind provided a strong contrast in terms of white spruce stem density. Even though the slope and ridge LFs are similar in distance to unburned edge (Table 2.1), the slope whole plot is oriented perpendicular to the prevailing wind direction in a burned finger bordered on three sides by unburned spruce-hardwood forest (Figure 2.1). By contrast, the ridge is parallel to the prevailing wind and less of its area is within the wind-dispersed seed shadow. As a result, the less favorable position of the ridge S/SPs with respect to unburned white spruce seed sources and prevailing winds appears to have contributed to lower average spruce density compared to the slope (Figure 2.7b).

Of course, in forest management the timing of salvage harvest is tied to the fire season and not the masting cycle per se. White spruce masting in Interior Alaska is both infrequent and irregular (Zasada and Viereck, 1970; Fox *et al.*, 1984). An average of 12 years has been reported between good seed crops (Juday *et al.*, 2003). Environmental factors that control the initiation, viability, and magnitude of periodic white spruce seed crops in Interior Alaska apparently maximize the probability that seeds, and especially mast crops, will be released into landscapes in which fires have recently occurred (Juday *et al.*, 2003; Roland *et al.*, 2014). Seedbed receptivity to white spruce seedling establishment is considerably improved by fire consumption of the upper organic layer of the forest floor (Galipeau *et al.*, 1997; Purdy *et al.*, 2002; Peters *et al.*, 2005). Established white spruce seedling density following fire is not necessarily explained by masting events only, because factors such as immediate post-fire weather and fire severity effects on the seedbed may also exert a controlling influence (Peters *et al.*, 2006). Nevertheless, as a result of all these factors, burned stands are likely to experience higher natural white spruce regeneration success than stands originating from arbitrarily scheduled harvest of green stands. The effects of enhanced natural spruce recruitment following the fire were present even in our site managed for assisted regeneration of white spruce, and natural regeneration may therefore be a robust regeneration strategy in post-fire salvage stands.

2.5.4.Birch

The experimental regeneration treatments of the RCFTRI were designed to increase white spruce establishment and early growth. However, all the factors we analyzed continued to exert a large influence on birch basal area and stem density 28 years after they were applied. Birch basal area and stem density appeared to be affected by environmental factors characteristic of each of the two landforms (ridge and slope). Birch dominated the ridge, contributing 84% of total basal area compared to only 44% on the slope. This difference in relative birch abundance may be related to differential hardwood tree recruitment on the two landforms. Birch establishment likely did not face a seedbed limitation, as the thick organic layers that typically inhibit birch recruitment (Densmore and Page, 1992) were greatly reduced by the combined effects of severe burning and salvage logging. Birch density and basal area also did not appear to be limited by the higher levels of *C. canadensis* on the ridge measured after initial treatment (Densmore *et al.*, 1999), even though birch stem density is inversely related to *C. canadensis* cover (Densmore and Page, 1992). Apparently, organic layer thickness on a burned and salvaged site exercises greater control over birch abundance than grass cover.

Differences in birch and aspen abundance on the two landforms may instead relate to the preference for different site types, as well as the legacy of pre-fire stand composition. Biomass of these two hardwood species in our study was significantly negatively correlated (Figure 2.5b), suggesting they tend not to flourish on the same sites. Upland boreal forest communities are sorted by hill position, which controls moisture and nutrient gradients as well as growing season solar radiation (Viereck *et al.*, 1986; Kurkowski *et al.*, 2008). On the RCFTRI, birch basal area was negatively correlated with cumulative solar radiation, while aspen was positively correlated (Figure or Table). In general in Interior Alaska, birch tolerates lower levels of insolation than aspen, and is more prevalent on cooler, moister sites (Chapin *et al.*, 2006a). Differential early success of birch versus aspen establishment can be reinforced through subsequent competition. On severely burned sites, small-seeded species such as birch are at a competitive disadvantage compared to resprouting aspen connected to an established legacy root system (Johnstone and Chapin, 2006). Birch regeneration is very sensitive to reduced light levels from vegetative competition (Perala and Alm, 1990), and overtopping by fast-growing aspen shoots can prove lethal (LaBonte and Nash, 1978). It appears that high levels of initial aspen abundance on the slope did, in fact, constrain birch recruitment from seed relative to the ridge.

In this study, S/SPs that were burned only were not as favorable for birch establishment and growth as sites that received scarification as well. Any type of scarification nearly doubled birch basal

area compared to non-scarified S/SPs, and intensive GSTs such as double-disc and blade produced the largest effects (Figure 2.4c). Birch recruitment, measured by stem density, was significantly increased by double disc and blade scarification (Figure 2.4c). However, the ground scarification factor was not significant in the overall ANOVA in relation to birch basal area and density (Table 2.6). When evaluating four or more treatment groups, if most of the groups are clustered near the mean except one or two outliers, an F-test used in the ANOVA may indicate that the factor had an insignificant effect on the mean. In such a case, a post hoc test (e.g., Dunnett) may still detect significant differences between the outlier and the other treatments (Cohen, 2004). When the GST factor across our study is evaluated for its effect on stem density, the blade treatment is the outlier while the other scarification methods are relatively clustered. We therefore conclude that the more intensive (double disc and blade) ground scarification treatments did in fact increase birch stem density and growth at RCFTRI.

Birch stem density was significantly negatively correlated with distance to nearest unburned edge. Birch seed are typically wind-dispersed, and peak dispersal occurs in the fall (Safford *et al.*, 1990). However, prediction of variation in birch stem density in our study was not improved by incorporating the prevailing wind direction. In Interior Alaska viable birch seeds have been recorded up to 200 m from the seed source irrespective of wind direction (Zasada, 1985), suggesting birch seed dispersal is less reliant on fall winds to reach suitable seedbeds than white spruce in our study area. Birch seeds continue to abscise from the parent at reduced rates throughout the winter and often travel across the snow surface to greater distance from parent trees than the initial landing point (Safford *et al.*, 1990). Birch can also reproduce asexually from root collar sprouting, though sprouting ability begins to decline after 55 years (Safford *et al.*, 1990). The burned stands that comprise the study area originated between the late 18th and early 19th centuries (Juday *et al.*, 2013), so asexual stump sprouting by birch was likely a minor contribution to birch recruitment in stands of this age. As a result, we believe that birch largely reproduced in our study area from seed, and that birch seedfall is not as distance-limited or wind-dependent as white spruce in our study area.

2.5.5.Aspen

Our statistical methods were sensitive to the highly variable presence of aspen and the resulting skewed data distribution. In order to make statistically robust inferences about this species in a silvicultural ecperiment such as RCFTRI, the experiment would need to be applied on post-fire sites that had a more even pre-fire aspen component (e.g., Paragi and Haggstrom, 2007). Nevertheless, we observed strong tendencies that are highly indicative of aspen's probable response to the experimental

treatments. Aspen was more prevalent on the slope LF, with approximately three times as much aspen basal area and stem density compared to the ridge. The interaction of the LF and GST terms with regard to aspen basal area and biomass likely derived from different relative abundance of aspen between slope and ridge. Aspen stem density was not correlated significantly with any seed dispersal measure, which suggests that many aspen on the study area sprouted asexually. Another alternative is that aspen's plumose seeds, which are capable of long-distance dispersal (Perala, 1990), may have uniformly distributed across the study site. Generally, sexual reproduction of aspen is less effective than asexual sprouting in Interior Alaska due to stringent mineral seedbed and microclimatic germination requirements of the seed (Johnstone and Chapin, 2006; Paragi and Haggstrom, 2007). Surveys after the Rosie Creek fire found hardwood self-replacement to be the norm, due in part to the early spring timing of the fire and the availability of unused root carbohydrate reserves at that time of year (Foote and Viereck, 1985). Within the burn scar of the Rosie Creek Fire, aspen regenerated in all pre-fire forest types, but regenerated aspen stem density was up to two orders of magnitude greater in pre-fire aspen stands (MacCracken and Viereck, 1990). Certain site types are more suitable to aspen growth due to earlier snow melt, higher solar radiation, and higher rates of evapotranspiration that disadvantage potential competitors (Chapin *et al.*, 2006a; Kurkowski *et al.*, 2008). Vigorous competition from birch and quick-growing shrub species has also been observed to reduce aspen sprouting densities (Paragi and Haggstrom, 2007). Site types inherently favorable to aspen often have high levels of aspen basal area that readily regenerate after disturbance. These adaptive mechanisms for aspen persistence suggest that when considering silvicultural regeneration treatments in stands dominated by aspen prior to disturbance, a significant legacy effect can be expected.

Intensive scarification reduced aspen basal area and stem density by half compared to non-scarified S/SPs on the slope whole plot (see 2.4.8). On green-harvested sites, reduction of thick moss layers or removal of slash increases soil warmth and stimulates aspen sprouting (Bella, 1986; Peterson and Peterson, 1995; Paragi and Haggstrom, 2007). However, scarification methods that cause deep disturbance to the soil and shallow aspen root systems are associated with reductions in aspen density and growth (Peterson and Peterson, 1995). Reductions in aspen biomass and suckering potential after harvest and scarification have also been attributed to soil compaction, specifically to changes in soil nutrient availability and porosity, in addition to the elimination of below-ground budbanks (Stone and Elioff, 1998; Haeussler and Kabzems, 2005). In northern Saskatchewan, disc and blade scarification reduced aspen biomass and stem density by over 70% compared to non-scarified sites, likely due to the elimination of regenerating shoots rather than decreasing the vigor of remnant shoots (Peltzer *et al.*,

2000). Vigorous competition from birch and quick-growing shrub species has also been observed to reduce aspen sprouting densities (Paragi and Haggstrom, 2007), and the positive response of birch establishment to increasing scarification intensity in our study may have further reduced aspen regeneration already weakened by scarification-induced soil disturbance. In evaluating our results at the RCFTRI, it is important to remember that the Rosie Creek Fire significantly reduced the organic layer before site preparation or regeneration treatments. As a result, we believe that lower aspen recruitment on S/SPs receiving an intensive GST was as a function of elimination of below-ground buds, compacted soil that reduced the growth potential of surviving buds, and vigorous competition. Aspen stem density was negatively related to the intensity of scarification, suggesting that seedbed was not a key limitation, and therefore the majority of reproduction was from clonal root suckering. Intensive scarification works against renewal of vigorous aspen stands in Interior Alaska.

2.6. Conclusion

This study contributes to our understanding of the durability of silvicultural treatments into the stem exclusion stage of forest development within the boreal forest in an experiment originally designed to test the effectiveness of multiple white spruce regeneration practices. Many studies in the North American boreal forest address the effect of silvicultural treatments on a single tree species, specifically white spruce in the great majority of cases. The experiment measured in this study provided an excellent opportunity to compare treatment effects on white spruce density and growth. The RCFTRI experiment represents an operational-scale, spatially-explicit, replicated design on a single site disturbed consecutively by high-severity wildfire and clearcut salvage harvest. All three of the experimental factors exercised continuing influence on the patterns of white spruce regeneration and growth. The treatment effects did not attenuate over time for white spruce, and we found statistically significant effects that the original researchers could only describe as tendencies. However, relatively few studies address treatment impacts on non-target species (e.g., Cole *et al.*, 1999; Wurtz and Zasada, 2001), or determine how the silvicultural treatments affect a site's net woody biomass production. Experimental silvicultural practices targeted in this study to improve white spruce survival had profound effects on other dominant tree species. Total biomass was increased on birch dominated areas that received scarification, but planting spruce merely shifted site biomass away from hardwoods largely into canopy-dominant spruce. Interior Alaska timber species demonstrate different regeneration strategies to post-disturbance environmental conditions, especially residual organic soil layer thickness and spatial configuration of surviving potential seed sources. Effective silvicultural practices must consider each

species' unique reproductive biology, and clonal sprouting as a source of aspen persistence was a particularly important example in our study. Site differences, such as we found between the slope and ridge landforms, are a key consideration for implementing effective silvicultural practices. Statistically significant interactions between the regeneration treatments and landform types proved to be critical to meet specific reforestation objectives, particularly the different herbaceous vegetation cover types, presence/absence of aspen clonal rootstocks, and spatial configurations regarding seed sources. Managing mixed species stands, which are common in the lightly managed portions of the boreal forest, requires not only the consideration of the future crop tree, but the interacting effects of silvicultural practices on all tree species. Regeneration silviculture, if it is to be efficient in meeting management goals, must take into account the likely effects of these different interactions. Future stand dynamics in the boreal forest are determined during a brief post-disturbance window, and reforestation silviculturalists should pay special focus to the influence of treatments in the manager's toolbox at the stand initiation stage.

2.7. References

AEA, 2011. Renewable energy atlas of Alaska. Alaska Energy Authority

AHAP, 1988. Vegetation classification for Bonanza Creek Experimental Forest. In, Alaska High-Altitude Aerial Photography Program GeoData Center, Geophysical Institute, University of Alaska Fairbanks, Fairbanks AK.

Alaska, S.o., 2010. SDMI IFSAR N64W148 DTM. In. Geographic Information Network of Alaska, University of Alaska Fairbanks.

Alden, J., 1985. Early surival and growth of white spruce on natural sites, in: (Eds.) Dyrness, C.T., Juday, G.P., Early Results of the Rosie Creek Fire Research Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK, pp. 40-43.

Awada, T., Henebry, G.M., Redmann, R.E., Sulistiyowati, H., 2004. Picea glauca dynamics and spatial pattern of seedlings regeneration along a chronosequence in the mixedwood section of the boreal forest. Ann. For. Sci. 61, 789-794. doi: 10.1051/forest:2004079.

Balshi, M.S., McGuire, A.D., Duffy, P., Flannigan, M., Walsh, J., Melillo, J., 2009. Assessing the response of area burned to changing climate in western boreal North America using a Multivariate Adaptive Regression Splines (MARS) approach. Glob. Change Biol. 15, 578-600. doi: 10.1111/j.1365-2486.2008.01679.x.

Barber, V.A., Juday, G.P., Finney, B.P., Wilmking, M., 2004. Reconstruction of summer temperatures in Interior Alaska from tree-ring proxies: evidence for changing synoptic climate regimes. Climatic Change 63, 91-120. doi: 10.1023/B:CLIM.0000018501.98266.55.

Beck, P.S.A., Juday, G.P., Alix, C., Barber, V.A., Winslow, S.E., Sousa, E.E., Heiser, P., Herriges, J.D., Goetz, S.J., 2011. Changes in forest productivity across Alaska consistent with biome shift. Ecol. Lett. 14, 373-379. doi: 10.1111/j.1461-0248.2011.01598.x.

Bedford, L., Sutton, R.F., Stordeur, L., Grismer, M., 2000. Establishing white spruce in the Boreal White and Black Spruce Zone. New Forests 20, 213-233. doi: 10.1023/A:1006774518199.

Beget, J.E., Stone, D., Verbyla, D.L., 2006. Regional overview of Interior Alaska, in: (Eds.) Chapin, F.S., Oswood, M.W., Cleve, K.V., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 12-20.

Bella, I.E., 1986. Logging practices and subsequent development of aspen stands in east-central Saskatchewan. Forestry Chron. 62, 81-83. doi: 10.5558/tfc62081-2.

Boateng, J.O., Heineman, J.L., Bedford, L., Harper, G.J., Linnell-Nemec, A.F., 2009. Long-term effects of site preparation and postplanting vegetation control on Picea glauca survival, growth and predicted yield in boreal British Columbia. Scand. J. For. Res. 24, 111-129. doi: 10.1080/02827580902759685.

Boateng, J.O., Heineman, J.L., McClarnon, J., Bedford, L., 2006. Twenty year responses of white spruce to mechanical site preparation and early chemical release in the boreal region of northeastern British Columbia. Can. J. For. Res. 36, 2386-2399. doi: 10.1139/x06-197.

Boucher, D., Gauthier, S., Noël, J., Greene, D.F., Bergeron, Y., 2014. Salvage logging affects early post-fire tree composition in Canadian boreal forest. For. Ecol. Manage. 325, 118-127. doi: 10.1016/j.foreco.2014.04.002.

Burnett, M.R., August, P.V., Brown, J.H., Jr., Killingbeck, K.T., 1998. The influence of geomorphological heterogeneity on biodiversity. Conservation Biol. 12, 363-370. doi: 10.1111/j.1523-1739.1998.96238.x.

Cater, T.C., Chapin, F.S., 2000. Differential Effects of Competition or Microenvironment on Boreal Tree Seedling. Ecology 81, 1086-1099. doi: 10.1890/0012-9658(2000)081[1086:DEOCOM]2.0.CO;2.

Chapin, F.S., Callaghan, T.V., Bergeron, Y., Fukuda, M., Johnstone, J.F., Juday, G.P., Zimov, S.A., 2004. Global change and the boreal forest: thresholds, shifting states or gradual change? Ambio 33, 361-365. doi: 10.1579/0044-7447-33.6.361.

Chapin, F.S., Hollingsworth, T.N., Murray, D.F., Viereck, L.A., Walker, M.D., 2006a. Floristic diversity and vegetation distribution in the Alaskan boreal forest, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 81-99.

Chapin, F.S., McGuire, A.D., Ruess, R.W., Hollingsworth, T.N., Mack, M.C., Johnstone, J.F., Kasischke, E.S., Euskirchen, E.S., Jones, J.B., Jorgenson, M.T., Kielland, K., Kofinas, G.P., Turetsky, M.R., Yarie, J., Lloyd, A.H., Taylor, D.L., 2010. Resilience of Alaska's boreal forest to climatic change. Can. J. For. Res. 40, 1360-1370. doi: 10.1139/X10-074.

Chapin, F.S., Viereck, L.A., Adams, P.C., Van Cleve, K., Fastie, C.L., Ott, R.A., Mann, D., Johnstone, J.F., 2006b. Successional processes in the Alaskan boreal forest, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 100-119.

Chen, H.Y.H., Popadiouk, R.V., 2002. Dynamics of North American boreal mixedwoods. Environ. Rev. 10, 137-166. doi: 10.1139/a02-007.

Cohen, B.H., 2004. Explaining Psychological Statistics 2nd Edition. Wiley, Hoboken NJ.

Cole, E., Youngblood, A., Newton, M., 2003. Effects of competing vegetation on juvenile white spruce (Picea glauca (Moench) Voss) growth in Alaska. Ann. For. Sci. 60, 573-583. doi: 10.1051/forest:2003049.

Cole, E.C., Newton, M., Youngblood, A., 1999. Regenerating white spruce, paper birch, and willow in south-central Alaska. Can. J. For. Res. 29, 993-1001. doi: 10.1139/x99-030.

Collins, W.B., Schwartz, C.C., 1998. Logging in Alaska's boreal forest: creation of grasslands or enhancement of moose habitat. Alces 34, 355-374. doi: N/A.

Comeau, P.G., Gendron, F., Letchford, T., 1998. A comparison of several methods for estimating light under a paper birch mixedwood stand. Can. J. For. Res. 28, 1843-1850. doi: 10.1139/x98-159.

Comeau, P.G., Kabzems, R., McClarnon, J., Heineman, J.L., 2005. Implications of selected approaches for regenerating and managing western boreal mixedwoods. Forestry Chron. 81, 559-574. doi: 10.5558/tfc81559-4.

Comeau, P.G., Wang, J.R., Letchford, T., 2003. Influences of paper birch competition on growth of understory white spruce and subalpine fir following spacing. Can. J. For. Res. 33, 1962-1973. doi: 10.1139/x03-117.

Cortini, F., Comeau, P.G., Boateng, J.O., Bedford, L., 2010. Yield implications of site preparation treatments for lodgepole pine and white spruce in northern British Columbia. Forests 1, 25-48. doi: 10.3390/f1010025.

Crimp, P.M., Phillips, S.J., Worum, G.T., 1997. Resources on State Forestry Lands in the Tanana Valley. Division of Forestry, Alaska Department of Natural Resources

Crow, T.R., Schlaegel, B.E., 1988. A guide to using regression equations for estimating tree biomass. North. J. Appl. Forestry 5, 15-22. doi: N/A.

DeLong, H.B., Lieffers, V.J., Blenis, P.V., 1997. Microsite effects on first-year establishment and overwinter survival of white spruce in aspen-dominated boreal mixedwoods. Can. J. For. Res. 27, 1452-1457. doi: 10.1139/x97-109.

Densmore, R.V., Juday, G.P., Zasada, J.C., 1999. Regeneration alternatives for upland white spruce after burning and logging in interior Alaska. Can. J. For. Res. 29, 413-423. doi: 10.1139/x99-008.

Densmore, R.V., Page, J.C., 1992. Paper birch regeneration on scarified logged areas in Southcentral Alaska. North. J. Appl. Forestry 9, 63-66. doi: N/A.

Deutschman, D.H., Levin, S.A., Pacala, S.W., 1999. Error propagation in a forest succession model: the role of fine-scale heterogeneity in light. Ecology 80, 1927-1943. doi: 10.1890/0012-9658(1999)080[1927:EPIAFS]2.0.CO;2.

DOF, 2007. Alaska Forest Resources & Practices Regulations. In. Division of Forestry, Alaska Department of Natural Resources, http://forestry.alaska.gov/pdfs/2009AFResourcesPracticesRegulations_June2007.pdf, p. 68.

DOF, 2010. Statewide assessment of forest resources. Division of Forestry, Alaska Department of Natural Resources

DOF, 2013. Final decision best interest finding for Mississippi Fire salvage. Division of Forestry, Alaska Department of Natural Resources

Eis, S., 1981. Effect of vegetative competition on regeneration of white spruce. Can. J. For. Res. 11, 1-8. doi: 10.1139/x81-001.

ESRI, 2012. ArcMap GIS. In. Enivronmental Systems Research Institute, Inc., Redlands, CA.

Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R., Stocks, B.J., 2005. Future area burned in Canada. Climatic Change 72, 1-16. doi: 10.1007/s10584-005-5935-y.

Foote, J., Viereck, L., 1985. Burn severity: its impact on the natural revegetation process following the Rosie Creek Fire, in: (Eds.) Dyrness, C.T., Juday, G.P., Early Results of the Rosie Creek Fire Research

Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK, pp. 26-29.

Fox, J.D., Zasada, J.C., Gasbarro, A.F., Van Veldhuizen, R., 1984. Monte Carlo simulation of white spruce regeneration after logging in interior Alaska. Can. J. For. Res. 14, 617-622. doi: 10.1139/x84-112.

Fresco, N., Chapin, F.S., 2009. Assessing the potential for conversion to biomass fuels in interior Alaska. U.S. Department of Agriculture, Forest Service PNW-RP-579.

Galipeau, C., Kneeshaw, D., Bergeron, Y., 1997. White spruce and balsam fir colonization of a site in the southeastern boreal forest as observed 68 years after fire. Can. J. For. Res. 27, 139-147. doi: 10.1139/x96-148.

Gallant, A.L., Binnian, E.F., Omernik, J.M., Shasby, M.B., 1995. Ecoregions of Alaska. US Geological Survey Professional Paper 1567.

Greene, D.F., Gauthier, S., Noël, J., Rousseau, M., Bergeron, Y., 2006. A field experiment to determine the effect of post-fire salvage on seedbeds and tree regeneration. Front. Ecol. Environ. 4, 69-74. doi: 10.1890/1540-9295(2006)004[0069:AFETDT]2.0.CO;2.

Greene, D.F., Macdonald, S.E., Haeussler, S., Domenicano, S., Noël, J., Jayen, K., Charron, I., Gauthier, S., Hunt, S., Gielau, E.T., Bergeron, Y., Swift, L., 2007. The reduction of organic-layer depth by wildfire in the North American boreal forest and its effect on tree recruitment by seed. Can. J. For. Res. 37, 1012-1023. doi: 10.1139/X06-245.

Greene, D.F., Zasada, J.C., Sirois, L., Kneeshaw, D., Morin, H., I. Charron, Simard, M.-J., 1999. A review of the regeneration dynamics of North American boreal forest tree species. Can. J. For. Res. 29, 824-839. doi: 10.1139/x98-112.

Haeussler, S., Kabzems, R., 2005. Aspen plant community response to organic matter removal and soil compaction. Can. J. For. Res. 35, 2030-2044. doi: 10.1139/x05-133.

Hammond, T.O., 1996. Mapping Interior Alaska forest types with satellite imagery. In, School of Agriculture and Land Resources Management. University of Alaska, Fairbanks, Fairbanks, AK, p. 136.

Haugen, R.K., Slaughter, C.W., Howe, K.E., Dingman, S.L., 1982. Hydrology and climatology of the Caribou-Poker Creeks Research Watershed, Alaska. Cold Regions Research and Engineering Laboratory (CRREL), US Army Corps of Engineers Report 82-26.

Hawkins, C.B.D., Steele, T.W., Letchford, T., 2006. The economics of site preparation and the impacts of current forest policy: evidence from central British Columbia. Can. J. For. Res. 36, 482-494. doi: 10.1139/X05-262.

Heinselman, M.L., 1981. Fire and succession in the conifer forests of northern North America, in: (Eds.) West, D.C., Shugart, H.H., Botkin, D.B., Forest Succession. Springer, New York, pp. 374-405.

Hermanns, J., 2013. Five year schedule of timber sales 2013-2017. Division of Forestry, Alaska Department of Natural Resources

Hinzman, L.D., Viereck, L.A., Adams, P.C., Romanovsky, V.E., Yoshikawa, K., 2006. Climate and permafrost dynamics of the Alaskan boreal forest, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 39-61.

Hogg, E.H., Lieffers, V.J., 1991. The impact of Calamagrostis canadensis on soil thermal regimes after logging in northern Alberta. Can. J. For. Res. 21, 387-394. doi: 10.1139/x91-048.

Huang, S., Price, D., Titus, S.J., 2000. Development of ecoregion-based height-diameter models for white spruce in boreal forests. For. Ecol. Manage. 129, 125-141. doi: 10.1016/S0378-1127(99)00151-6.

Huang, S., Titus, S.J., 1994. An age-independent individual tree height prediction model for boreal spruce-aspen stands in Alberta. Can. J. For. Res. 24, 1295–1301. doi: 10.1139/x94-169.

Hurlbert, S.H., 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54, 187-211. doi: 10.2307/1942661.

Jenkins, J.C., Chojnacky, D.C., Heath, L.S., Birdsey, R.A., 2003. National-scale biomass estimators for United States tree species. For. Sci. 49, 12-35. doi: N/A.

Johnstone, J.F., Chapin, F.S., 2006. Effects of soil burn severity on post-fire tree recruitment in boreal forest. Ecosystems 9, 14-31. doi: 10.1007/s10021-004-0042-x.

Johnstone, J.F., Chapin, F.S., Foote, J., Kemmett, S., Price, K., Viereck, L., 2004. Decadal observations of tree regeneration following fire in boreal forests. Can. J. For. Res. 34, 267-273. doi: 10.1139/X03-183.

Johnstone, J.F., Hollingsworth, T.N., Chapin, F.S., Mack, M.C., 2010. Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. Glob. Change Biol. 16, 1281-1295. doi: 10.1111/j.1365-2486.2009.02051.x.

Johnstone, J.F., Rupp, T.S., Olson, M., Verbyla, D., 2011. Modeling impacts of fire severity on successional trajectories and future fire behavior in Alaskan boreal forests. Landsc. Ecol. 26, 487-500. doi: 10.1007/s10980-011-9574-6.

Juday, G.P., 1985. Preface: the Rosie Creek Fire and its research opportunities, in: (Eds.) Juday, G.P., Dyrness, C.T., Early Results of the Rosie Creek Fire Research Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK.

Juday, G.P., Barber, V., Rupp, T.S., Zasada, J.C., Wilmking, M., 2003. A 200-year perspective of climate variability and the response of white spruce in Interior Alaska, in: (Eds.) Greenland, D., Goodin, D.G., Smith, R.C., Climate variability and ecosystem response at long-term ecological research sites. Oxford University Press, New York, NY, pp. 226-250.

Juday, G.P., Densmore, R.V., Zasada, J.C., 2013. White spruce regeneration silviculture techniques 25 years after wildfire: the Rosie Creek Fire Tree Regeneration Installation., in: (Eds.) Camp, A.E., Irland, L.C., Carroll, C.J.W., Long-term Silvicultural & Ecological Studies: Results for Science and Management, Volume 2. Yale University School of Forestry and Environmental Studies, pp. 49-65.

Kasischke, E.S., Rupp, T.S., Verbyla, D.L., 2006. Fire trends in the Alaskan boreal forest, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 285-301.

Kasischke, E.S., Turetsky, M.R., 2006. Recent changes in the fire regime across the North American boreal region: spatial and temporal patterns of burning across Canada and Alaska. Geophysical Res. Lett. 33, 1-5. doi: 10.1029/2006GL025677.

Kasischke, E.S., Verbyla, D.L., Rupp, T.S., A. David McGuire, Murphy, K.A., Jandt, R., Barnes, J.L., Hoy, E.E., Paul A. Duffy, Calef, M., Turetsky, M.R., 2010. Alaska's changing fire regime - implications for the vulnerability of its boreal forests. Can. J. For. Res. 40, 1313-1324. doi: 10.1139/X10-098.

Kuehl, R.O., 2000. Design of experiments: statistical principles of research design and analysis. Duxbury Thomson Learning, Pacific Grove, CA.

Kurkowski, T.A., Mann, D.H., Rupp, T.S., Verbyla, D.L., 2008. Relative importance of different secondary successional pathways in an Alaskan boreal forest. Can. J. For. Res. 38, 1911-1923. doi: 10.1139/X08-039.

Kurulok, S., Macdonald, E., 2004. Impacts of post-fire salvage logging on tree regeneration and plant communities in the mixedwood boreal forest of Alberta Sustainable Forest Management Network

LaBonte, G.A., Nash, R.W., 1978. Cleaning and weeding paper birch -- a 24-year case history. Forestry 4, 223-225. doi: N/A.

Lechowicz, M.J., Bell, G., 1991. The ecology and genetics of fitness in forest plants. II. Microspatial heterogeneity of the edaphic environment. Ecology 79, 687-696. doi: N/A.

Lieffers, V.J., Macdonald, S.E., Hogg, E.H., 1993. Ecology of and control strategies for Calamagrostis canadensis in boreal forest sites. Can. J. For. Res. 23, 2070-2077. doi: 10.1139/cjfr-23-10-2070.

Lieffers, V.J., Stadt, K.J., Navratil, S., 1996. Age structure and growth of understory white spruce under aspen. Can. J. For. Res. 26, 1002-1007. doi: 10.1139/x26-110.

Lindenmayer, D.B., Burton, P.J., Franklin, J.F., 2008. Salvage Logging and its Ecological Consequences. Island Press, Washington, DC.

Lindenmayer, D.B., Foster, D.R., Franklin, J.F., Hunter, M.L., Noss, R.F., Schmiegelow, F.A., Perry, D., 2004. Salvage harvesting policies after natural disturbance. Science 303, 1303. doi: N/A.

MacCracken, J.G., Viereck, L.A., 1990. Browse regrowth and use by moose after fire in Interior Alaska. Northwest Sci. 64, 11-18. doi: N/A.

MacIsaac, D.A., Navratil, S., 1995. Competition dynamics in juvenile boreal hardwood-conifer mixtures. In: Comeau, P.G., Thomas, K.D. (Eds.), Silviculture of temperate and boreal broadleaf-conifer mixtures. BC Ministry of Forests, Richmond, BC, pp. 23-34.

Malone, T., Liang, J., Packee, E.C., 2009. Cooperative Alaska forest inventory. U.S. Department of Agriculture, Forest Service PNW-GTR-785.

Man, R., Lieffers, V.J., 1999. Are mixtures of aspen and white spruce more productive than single species stands? Forestry Chron. 75, 505-513. doi: 10.5558/tfc75505-3.

Martin-DeMoor, J., Lieffers, V.J., Macdonald, S.E., 2010. Natural regeneration of white spruce in aspen-dominated boreal mixedwoods following harvesting. Can. J. For. Res. 40, 585-594. doi: 10.1139/X10-016.

Nappi, A., 2011. Harvesting in burned forests — issues and orientations for ecosystem-based management. Ministère des Ressources naturelles et de la Faune, Direction de l'environnement et de la protection des forêts

Nappi, A., Drapeau, P., Savard, J.-P.L., 2004. Salvage logging after wildfire in the boreal forest: Is it becoming a hot issue for wildlife? Forestry Chron. 80, 67-74. doi: 10.5558/tfc80067-1.

Oliver, C.D., Larson, B.C., 1996. Forest stand dynamics. Wiley, New York.

Paragi, T.F., Haggstrom, D.A., 2007. Short-term responses of aspen to fire and mechanical treatments in Interior Alaska. North. J. Appl. Forestry 24, 153-157. doi: N/A.

Peltzer, D.A., Bast, M.L., Wilson, S.D., Gerry, A.K., 2000. Plant diversity and tree responses following contrasting disturbances in boreal forest. For. Ecol. Manage. 127, 191-203. doi: 10.1016/S0378-1127(99)00130-9.

Perala, D.A., 1990. Populus tremuloides Michx. quaking aspen, in: (Eds.) Burns, R.M., Honkala, B.H., Silvics of North America: Volume 2, Hardwoods. U.S. Department of Agriculture, Forest Service, Washington DC, p. 34.

Perala, D.A., Alm, A.A., 1990. Regeneration silviculture of birch: a review. For. Ecol. Manage. 32, 39-77. doi: 10.1016/0378-1127(90)90105-K.

Peters, V.S., Macdonald, S.E., Dale, M.R.T., 2005. The interaction between masting and fire is key to white spruce regeneration. Ecology 86, 1744-1750. doi: 10.1890/03-0656.

Peters, V.S., Macdonald, S.E., Dale, M.R.T., 2006. Patterns of initial versus delayed regeneration of white spruce in boreal mixedwood succession. Can. J. For. Res. 36, 1597-1609. doi: 10.1139/X06-020.

Peterson, N.M., Peterson, E.B., 1995. Aspen Managers' Handbook for British Columbia. Pacific Forestry Centre, Canadian Forest Service, Natural Resources Canada

Pillsbury, N.H., 1995. Reliability of tanoak volume equations when applied to different areas West. J. Appl. Forestry 10, 72-78. doi: N/A.

Pitt, D.G., Comeau, P.G., Parker, W.C., MacIsaac, D., McPherson, S., Hoepting, M.K., Stinson, A., Mihajlovich, M., 2010. Early vegetation control for the regeneration of a single-cohort, intimate mixture of white spruce and trembling aspen on upland boreal sites. Can. J. For. Res. 40, 549-564. doi: 10.1139/X10-012.

Puettmann, K.J., Coates, K.D., Messier, C., 2009. A Critique of Silviculture. Island Press, Washington DC.

Purdy, B.G., Macdonald, S.E., Dale, M.R.T., 2002. The regeneration niche of white spruce following fire in the mixedwood boreal forest. Silva Fennica 36, 289-306. doi: 10.14214/sf.564.

Putman, W.E., Zasada, J.C., 1986. Direct seeding techniques to regenerate white spruce in interior Alaska. Can. J. For. Res. 16, 660-664. doi: 10.1139/x86-115.

Reed, R.A., Peet, R.K., Palmer, M.W., White, P.S., 1993. Scale dependence of vegetation-environment correlations: A case study of a North Carolina piedmont woodland. Veg. Sci. 4, 329-340. doi: 10.2307/3235591.

Roland, C.A., Schmidt, J.H., Johnstone, J.F., 2014. Climate sensitivity of reproduction in a mast-seeding boreal conifer across its distributional range from lowland to treeline forests. Oecologia 174, 665-677. doi: 10.1007/s00442-013-2821-6.

Rupp, T.S., 1997. A geographic model of landscape-level post-disturbance forest establishment patterns of Interior Alaska white spruce ecosystems. In, School of Natural Resources. University of Alaska, Fairbanks.

Safford, L.O., Bjorkbom, J.C., Zasada, J.C., 1990. Betula papyrifera Marsh. paper birch, in: (Eds.) Burns, R.M., Honkala, B.H., Silvics of North America: Volume 2, Hardwoods. U.S. Department of Agriculture, Forest Service, Washington DC, p. 19.

Saint-Germain, M., Greene, D.F., 2009. Salvage logging in the boreal and cordilleran forests of Canada: Integrating industrial and ecological concerns in management plans. Forestry Chron. 85, 120-134. doi: 10.5558/tfc85120-1.

SAS, 2012. SAS Software for Windows, Release 9.3. In. SAS Institute, Inc., Cary, NC.

Sharma, M., Zhang, S.Y., 2004. Height/diameter models using stand characteristics for Pinus banksiana and Picea mariana. Scand. J. For. Res. 19, 442-451. doi: 10.1080/02827580410030163.

Soil Survey Staff, N.R.C.S., U.S. Department of Agriculture, 2011. Web Soil Survey. In, http://websoilsurvey.nrcs.usda.gov/.

Stiell, W.M., 1976. White spruce: artificial regeneration in Canada. Forest Management Institute, Canadian Forestry Service, Department of the Environment

Stone, D.M., Elioff, J.D., 1998. Soil properties and aspen development five years after compaction and forest floor removal. Can. J. Soil Sci. 78, 51-58. doi: 10.4141/S97-026.

Sutton, R.F., 1995. White spruce establishment: initial fertilization, weed control, and irrigation evaluated after three decades. New Forests 9, 123-133. doi: 10.1007/BF00028685.

Thiffault, N., Jobidon, R., Munson, A.D., 2003. Performance and physiology of large containerized and bare-root spruce seedlings in relation to scarification and competition in Québec (Canada). Ann. For. Sci. 60, 645-655. doi: 10.1051/forest:2003057.

Tilman, D., Reich, P.B., Knops, J., Wedin, D., Mielke, T., Lehman, C., 2001. Diversity and productivity in a long-term grassland experiment. Science 26, 843-845. doi: 10.1126/science.1060391.

USGS, 1986. 22 August 1986 Color Infrared Photo: ID AR5860035821810. In: Earth Resources Observation and Science Center, U.S.G.S. (Ed.), Sioux Falls, SD.

Van Cleve, K., Chapin, F.S., Dyrness, C.T., Viereck, L.A., 1991. Element cycling in taiga forests: state-factor control. Bioscience 41, 78-88. doi: N/A.

Van Cleve, K., Chapin, F.S., Ruess, R.W., 2013a. Bonanza Creek Experimental Forest: Hourly Precipitation data, 1988 to Present. In. Bonanza Creek LTER - University of Alaska Fairbanks.

Van Cleve, K., Chapin, F.S., Ruess, R.W., 2013b. Bonanza Creek Experimental Forest: HourlyTemperature (sample, min, max) at 50 cm and 150 cm from 1988 to Present. In. Bonanza Creek LTER - University of Alaska Fairbanks.

Van Cleve, K., Dyrness, C.T., Viereck, L.A., Fox, J., Chapin, F.S., Oechel, W., 1983. Taiga ecosystems in Interior Alaska. Bioscience 33, 39-44. doi: 10.2307/1309243.

Viereck, L.A., Dyrness, C.T., Van Cleve, K., Foote, M.J., 1983. Vegetation, soils, and forest productivity in selected forest types in interior Alaska. Can. J. For. Res. 13, 703-720. doi: 10.1139/x83-101.

Viereck, L.A., Slaughter, C.W., 1986. Climatic characteristics of the taiga in Interior Alaska, in: (Eds.) Van Cleve, K., Chapin, F.S., Flanagan, P.W., Viereck, L.A., Dyrness, C.T., Forest Ecosystems in the Alaskan Taiga. Springer Verlag, New York, pp. 9-21.

Viereck, L.A., Van Cleve, K., Dyrness, C.T., 1986. Forest ecosystem distribution in the taiga environment, in: (Eds.) Van Cleve, K., Chapin, F.S., Flanagan, P.W., Viereck, L.A., Dyrness, C.T., Forest Ecosystems in the Alaskan Taiga. Springer Verlag, New York, pp. 22-43.

Weber, M.G., Flannigan, M.D., 1997. Canadian boreal forest ecosystem structure and function in a changing climate: impact on fire regimes. Environ. Rev. 5, 145-166. doi: 10.1139/a97-008.

Wiensczyk, A., Swift, K., Morneault, A., Thiffault, N., Szuba, K., Bell, F.W., 2011. An overview of the efficacy of vegetation management alternatives for conifer regeneration in boreal forests. Forestry Chron. 87, 175-200. doi: 10.5558/tfc2011-007.

Wurtz, T.L., 1995. Understory alder in three boreal forests of Alaska: local distribution and effects on soil fertility. Can. J. For. Res. 25. doi: 10.1139/x95-107.

Wurtz, T.L., Ott, R.A., Maisch, J.C., 2006. Timber harvest in Interior Alaska, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 302-308.

Wurtz, T.L., Zasada, J.C., 2001. An alternative to clear-cutting in the boreal forest of Alaska: a 27-year study of regeneration after shelterwood harvesting. Can. J. For. Res. 31, 999-1011. doi: 10.1139/cjfr-31-6-999.

Yarie, J., Van Cleve, K., 2006. Controls over forest production in Interior Alaska, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 171-188.

Yarie, J.A., Kane, E., Mack, M., 2007. Aboveground biomass equations for the trees of Interior Alaska. Agricultural and Forestry Experiment Station, University of Alaska Fairbanks

Youngblood, A., 1995. Development patterns in young conifer-hardwood forests of interior Alaska. Veg. Sci. 6, 229-236. doi: 10.2307/3236218.

Youngblood, A., Cole, E., Newton, M., 2011. Survival and growth response of white spruce stock types to site preparation in Alaska. Can. J. For. Res. 41, 793-809. doi: 10.1139/X11-001.

Youngblood, A., Max, T.A., 1992. Dispersal of white spruce seed on Willow Island in Interior Alaska. U.S. Department of Agriculture, Forest Service PNW-RP-443.

Youngblood, A., Zasada, J.C., 1991. White spruce artificial regeneration options on river floodplains in interior Alaska. Can. J. For. Res. 21, 423-433. doi: 10.1139/x91-057.

Zasada, J.C., 1976. Alaska's interior forests. Part I: Ecological silvicultural considerations. Forestry 74, 334-337. doi: N/A.

Zasada, J.C., 1980. Some considerations in the natural regeneration of white spruce in interior Alaska In: Murray, M., Van Veldhuizen, R.M. (Eds.), Forest Regeneration at High Latitudes. Pacific Northwest Forest and Range Experiment Station, U.S. Department of Agriculture, Forest Service, Fairbanks, AK, p. 52.

Zasada, J.C., 1985. Production, dispersal, and germination of white spruce and paper birch and first year seedling establishment after the Rosie Creek Fire, in: (Eds.) Juday, G.P., Dyrness, C.T., Early Results of the Rosie Creek Fire Research Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK, pp. 34-37.

Zasada, J.C., 1986. Natural regeneration of trees and tall Shrubs on forest sites in Interior Alaska, in: (Eds.) Van Cleve, K., Chapin, F.S., Flanagan, P.W., Viereck, L.A., Dyrness, C.T., Forest Ecosystems in the Alaskan Taiga. Springer Verlag, New York, pp. 44-73.

Zasada, J.C., Grigal, D., 1978. The effects of silvicutural system and seedbed preparation on natural regeneration of white spruce and associated species. In, Proceedings of the Fifth North American Forest Biology Workshop, 1978, Gainesville, Fla. Southern Forest Experiment Station, U.S. Department of Agriculture, Forest Service, Gainesville, FL, pp. 213-220.

Zasada, J.C., Norum, R., Van Veldhuizen, R.M., Tuetsch, C.E., 1983. Artificial regeneration of trees and tall shrubs in experimentally burned upland black spruce/feather moss stands in Alaska. Can. J. For. Res. 13, 903-913. doi: 10.1139/x83-120.

Zasada, J.C., Packee, E.C., 1995. The Alaska Region, in: (Ed.) Barrett, J.W., Regional Silviculture of the United States. Wiley, New York, pp. 559-606.

Zasada, J.C., Viereck, L.A., 1970. White spruce cone and seed production in interior Alaska, 1957-68. U.S. Department of Agriculture, Forest Service PNW-RN-129.

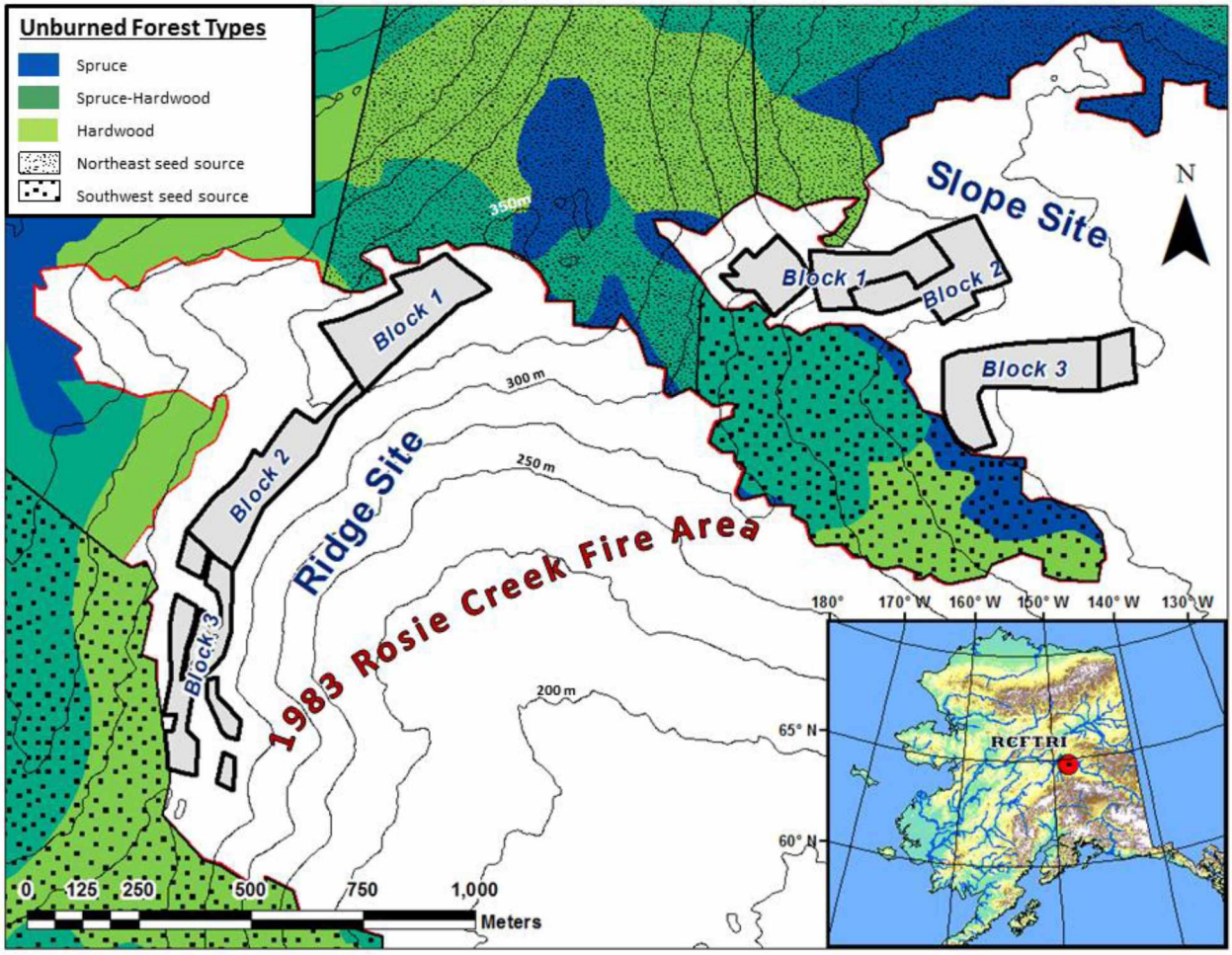

Figure 2.1: Overview map of RCFTRI with unburned forest types and GIS-classified wind-dispersed seed sources. Pre-fire forest types classified by AHAP (1988). Seed source regions and burned area classified by authors using ArcMap 10.1 (ESRI, Redlands, CA). Prevailing wind during fall seed abscission comes with equal likelihood from the northeast and southwest (Youngblood and Max, 1992).

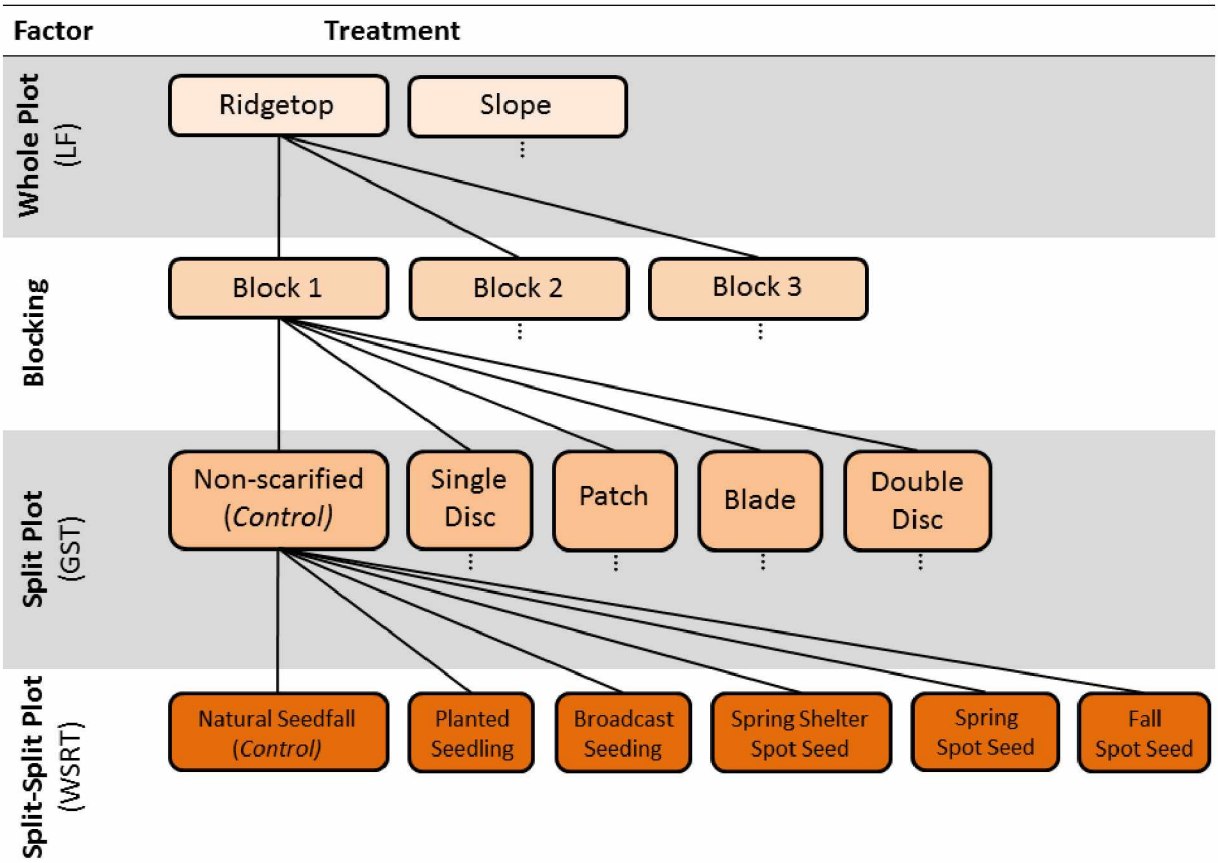

Figure 2.2: RCFTRI experimental design, consisting of four hierarchical levels organized in a blocked split-split plot design. LF = landform, GST = ground scarification treatment, WSRT = white spruce regeneration treatment. Each ellipsis represents an identical array of nested treatments as treatments at the same level.

Table 2.1: Summary of landform (LF) whole plot characteristics across 180 S/SPs. The slope and ridge LFs represent a number of topographical gradients that typify Alaska's upland boreal forest. Standard deviation indicated in parentheses.

	Ridge	Slope	Whole Experiment
Area	13.3 ha	13.4 ha	26.7 ha
Prefire Forest Type (AHAP, 1988)	White Spruce-Birch-Aspen, Birch-Aspen	White Spruce-Black Spruce-Birch-Aspen	Mixed spruce-Hardwood
Soil Type (Soil Survey Staff, 2011)	Gilmore Silt Loam 3-7% slope	Minto-Chatanika Complex 3-7% slope, Steese Silt Loam 20-30% slope	mod.-deep silt loam
Distance from Fire Edge	119.2 m *(49.4)*	151.6 m *(85.9)*	135.4 m
Southwest Seed Shadow Mean distance from wind-dispersed seed source	408.6 m *(306.7)*	163.7 m *(85.9)*	286.1 m
Northeast Seed Shadow Mean distance from wind-dispersed seed source	482.8 m *(342.8)*	297.7 m *(169.0)*	390.2 m

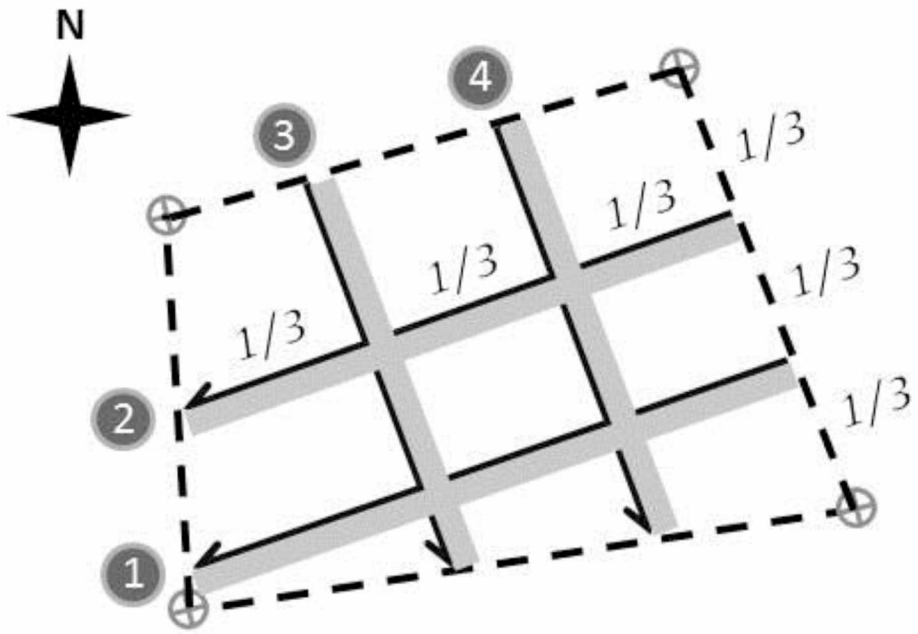

Figure 2.3: Depiction of typical belt transect layout. This layout procedure provided a representative sample of the entire S/SP.

Table 2.2: P-values of census and transect estimates by ten response variables.

	Stem Density	Basal Area
All species	0.583	0.757
Alder	0.082	0.095
Aspen	0.439	0.186
Birch	0.736	0.534
White Spruce	0.431	0.667

Table 2.3: Average height (m) by species and experimental factor, averaged from 14 S/SPs.

Experimental Factor	Aspen	Birch	White Spruce	Number of S/SPs
LF				
Ridgetop	12.40	8.00	3.73	6
Slope	10.61	7.05	5.23	8
GST				
Scarified	11.58	7.97	4.59	6
Non-Scarified	11.37	7.07	4.58	8
WSRT				
Planted Seedling	10.46	7.06	6.38	5
Broadcast	12.86	7.39	2.66	3
Fall Spot Seed	--	9.56	3.50	1
Control	11.31	7.47	4.17	5
Total	**11.42**	**7.46**	**4.59**	**14**

Table 2.4: Comparison of full and reduced regression models for height-diameter relationships by experimental factors; test statistics reported for interaction of experimental indicator variable and diameter. For the 'diameter only' row, test statistics are for diameter. ndf = numerator degrees of freedom, ddf = denominator degrees of freedom, and RMSE = root mean squared error.

* significant result ($\alpha<0.05$)

Experimental Factor	**Aspen**				**Birch**				**White Spruce**			
	ndf/ ddf	*F*	R^2	*RMSE*	*ndf/ ddf*	*F*	R^2	*RMSE*	*ndf/ ddf*	*F*	R^2	*RMSE*
LF												
Ridge vs Slope	1/91	0.99	0.77	155.688	1/314	**4.22***	0.88	113.584	1/202	0.25	0.93	0.138
GST												
Scarified vs Non-scarified	1/109	0.11	0.81	146.567	1/314	**5.30***	0.88	114.734	1/202	0.05	0.93	0.138
WSRT												
Plant vs NatSeed	1/78	0.40	0.75	152.480	1/196	3.38	0.89	110.245	1/159	2.70	0.92	0.142
Plant vs NatSeed vs Broadcast	2/89	0.90	0.77	156.27	2/264	1.76	0.88	113.623	2/190	1.78	0.93	0.134
Diameter only	1/111	**407.58***	0.79	155.207	1/316	**2128.69***	0.87	119.441	1/204	**2645.65***	0.93	0.138

Table 2.5: P-values for the linear relationship between distance to seed source and stem density, by species. White spruce figures calculated only with the subset of S/SPs receiving natural seedfall WSRT (n=30). Birch and aspen calculated with the 3WSRT dataset (n=90). All distances log-transformed except for mean distance to wind-dispersed seed source. (R^2 in parentheses)

* significant result ($\alpha<0.05$)

Species	Ln(Distance to Nearest Unburned Forest)	Mean Distance to Wind-Dispersed Seed Source	Ln(Distance to Nearest Wind-Dispersed Seed Source)	Ln(Distance to Nearest Wind-Dispersed Spruce Source)
White Spruce	0.6661	**0.0200***	0.0789	**0.0056***
	(0.01)	(0.18)	(0.11)	(0.24)
Birch	**0.0001***	**0.0235***	**0.0023***	n/a
	(0.23)	(0.06)	(0.10)	
Aspen	0.0518	0.1258	0.2371	n/a
	(0.04)	(0.03)	(0.02)	

Table 2.6: ANOVA table for the balanced 3WSRT dataset. LF = landform whole plot factor, GST = ground scarification split plot factor, and WSRT = white spruce regeneration split-split plot factor.

*significant result (α<0.05)

		All Species				White Spruce			
		biomass		stem density		basal area		stem density	
	df	F	p	F	p	F	p	F	p
LF	1	12.09	0.0737	0.66	0.5014	4.48	0.1686	10.10	0.0864
GST	4	4.15	**0.0171***	2.65	0.0719	0.52	0.7247	0.16	0.9541
GST*LF	4	0.34	0.8447	0.67	0.6207	1.63	0.2157	1.42	0.2732
WSRT	2	0.40	0.6714	3.99	**0.0262***	69.61	**0.0001***	21.61	**0.0001***
WSRT*LF	2	0.89	0.4170	1.11	0.3403	3.85	**0.0295***	6.16	**0.0046***
WSRT*GST	8	1.03	0.4304	0.35	0.9423	0.39	0.9188	0.44	0.8922
WSRT*LF*GST	8	1.49	0.1899	0.52	0.8330	1.06	0.4125	1.13	0.3635

		Birch				Aspen					
		basal area		stem density		basal area		stem density		biomass	
	df	F	p	F	p	F	p	F	p	F	p
LF	1	71.82	**0.0136***	5.51	0.1435	9.62	0.0901	13.2	0.0681	7.27	0.1145
GST	4	2.05	0.1358	2.15	0.1215	0.82	0.5335	2.06	0.1336	2.3	0.1034
GST*LF	4	0.98	0.4476	0.78	0.5526	2.46	0.0875	3.53	**0.0300***	3.28	**0.0384***
WSRT	2	3.35	**0.0450***	1.45	0.2476	0.8	0.4546	0.05	0.9554	0.19	0.8259
WSRT*LF	2	0.41	0.6693	1.52	0.2314	0.65	0.5280	0.55	0.5799	0.53	0.5953
WSRT*GST	8	2.49	**0.0271***	1.69	0.1299	1.28	0.2815	0.76	0.6381	0.51	0.8441
WSRT*LF*GST	8	1.34	0.2543	0.79	0.6111	0.81	0.5975	0.74	0.6559	0.73	0.6638

Table 2.7: Linear contrast results for all response variables using the 3WSRT dataset. Group A corresponds to the first category listed in the contrast name, Group B the second. Where applicable, references that prompted a contrast are given. All values reported as mean ± 1 SE.
'scar' = all scarification treatments, 'nonscar' = non-scarified control treatment, 'intscar' = intensive scarification treatments (i.e., blade and double-disc), 'plant' = planted seedling WSRT, 'natseed' = natural seedfall control WSRT, 'others' = two WSRTs not named in group A.
* Bonferroni-adjusted significant result (α=0.05/2=0.025)

Response variable	Contrast (group A vs group B)	Reference	F	p	Group A	Group B
Total biomass (kg ha^{-1})	scar vs nonscar		26.63	0.0001*	80,607±1,972 n=72	61,890±3,623 n=18
	plant vs others	Man and Lieffers (1999)	0.59	0.4465	78,303±2,849 n=30	76,144±2,481 n=60
Total density (stems ha^{-1})	scar vs nonscar		26.42	0.0001*	9,684±473 n=72	6,415±739 n=18
	broadcast vs others	Densmore *et al.* (1999)	7.24	0.0103*	9,942±673 n=30	8,574±540 n=60
White spruce basal area (m^2 ha^{-1})	scar vs nonscar		1.16	0.2871	3.79±0.54 n=72	3.80±0.81 n=18
	plant: intscar vs nonscar	Boateng *et al.* (2009)	0.83	0.3672	7.22±1.49 n=12	7.80±1.01 n=6
White spruce density (stems ha^{-1})	scar vs nonscar	DeLong *et al.* (1997)	0.18	0.6708	1,712±204 n=72	1,445±257 n=18
	natseed: scar vs nonscar	Rupp (1997)	0.12	0.7312	916±177 n=24	1,055±656 n=6
Birch basal area (m^2 ha^{-1})	scar vs nonscar	Zasada (1980)	45.04	0.0001*	13.29±0.92 n=72	7.60±0.72 n=18
	plant: scar vs nonscar		23.34	0.0001*	12.02±1.47 n=24	5.58±2.18 n=6
Birch density (stems ha^{-1})	scar vs nonscar	Zasada *et al.* (1983)	104.39	0.0001*	6,454±537 n=72	3,262±679 n=18
	plant vs others		2.31	0.1364	5,438±816 n=30	6,014±785 n=60
Aspen basal area (m^2 ha^{-1})	scar vs nonscar		2.10	0.1549	4.24±0.85 n=72	4.72±1.40 n=18
	intscar vs nonscar	Peltzer *et al.* (2000)	4.95	0.0317	2.84±0.91 n=36	4.72±1.40 n=18
Aspen biomass (kg ha^{-1})	scar vs nonscar		5.13	0.0291	14,837±2,877 n=72	17,377±4,986 n=18
	intscar vs nonscar	Peltzer *et al.* (2000)	8.01	0.0072*	10,002±3,110 n=36	17,377±4,986 n=18
Aspen density (stems ha^{-1})	scar vs nonscar	Stone and Elioff (1998)	4.20	0.0471	687±162 n=72	647±228 n=18
	intscar vs nonscar	Peltzer *et al.* (2000)	7.12	0.0109*	437±178 n=36	647±228 n=18

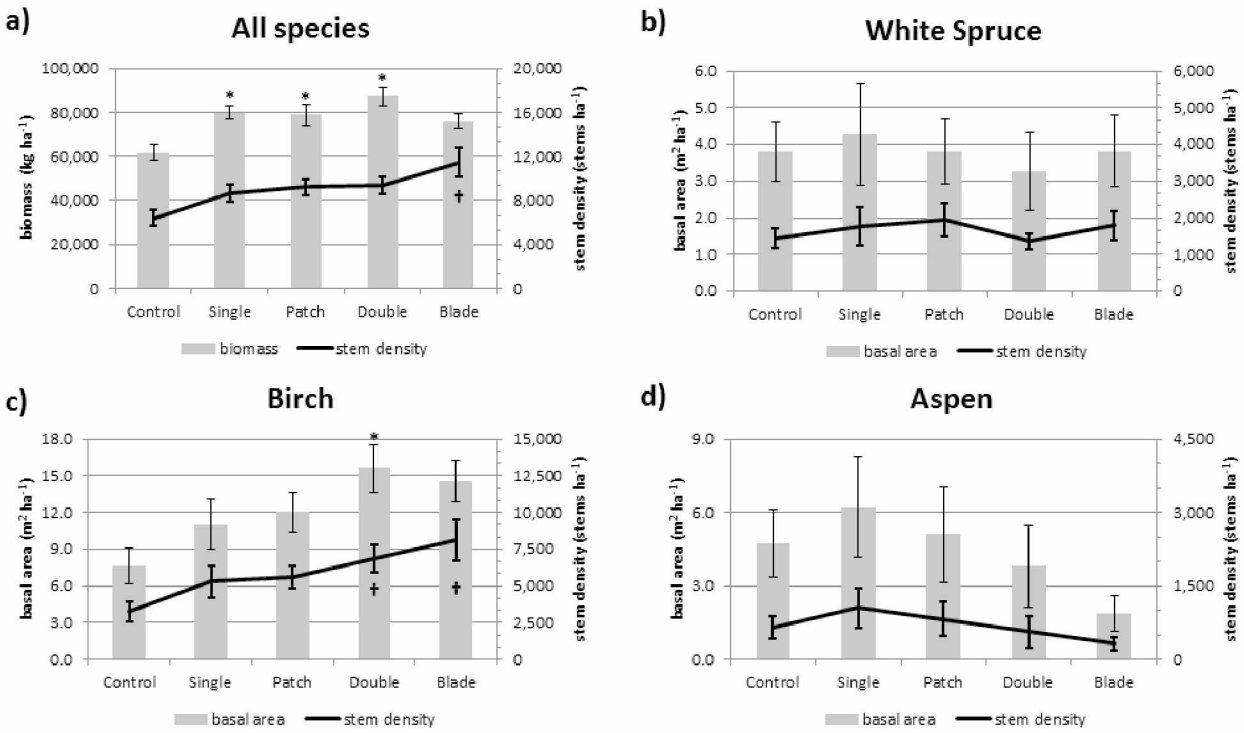

Figure 2.4: Ground scarification treatment (GST) group means for all species (a), white spruce (b), birch (c), and aspen (d); figures from the 3WSRT dataset. Error bars are ± 1 SE.

* Basal area significantly different from non-scarified control using Dunnett ($p<0.05$)

† Stem density significantly different from non-scarified control using Dunnett ($p<0.05$)

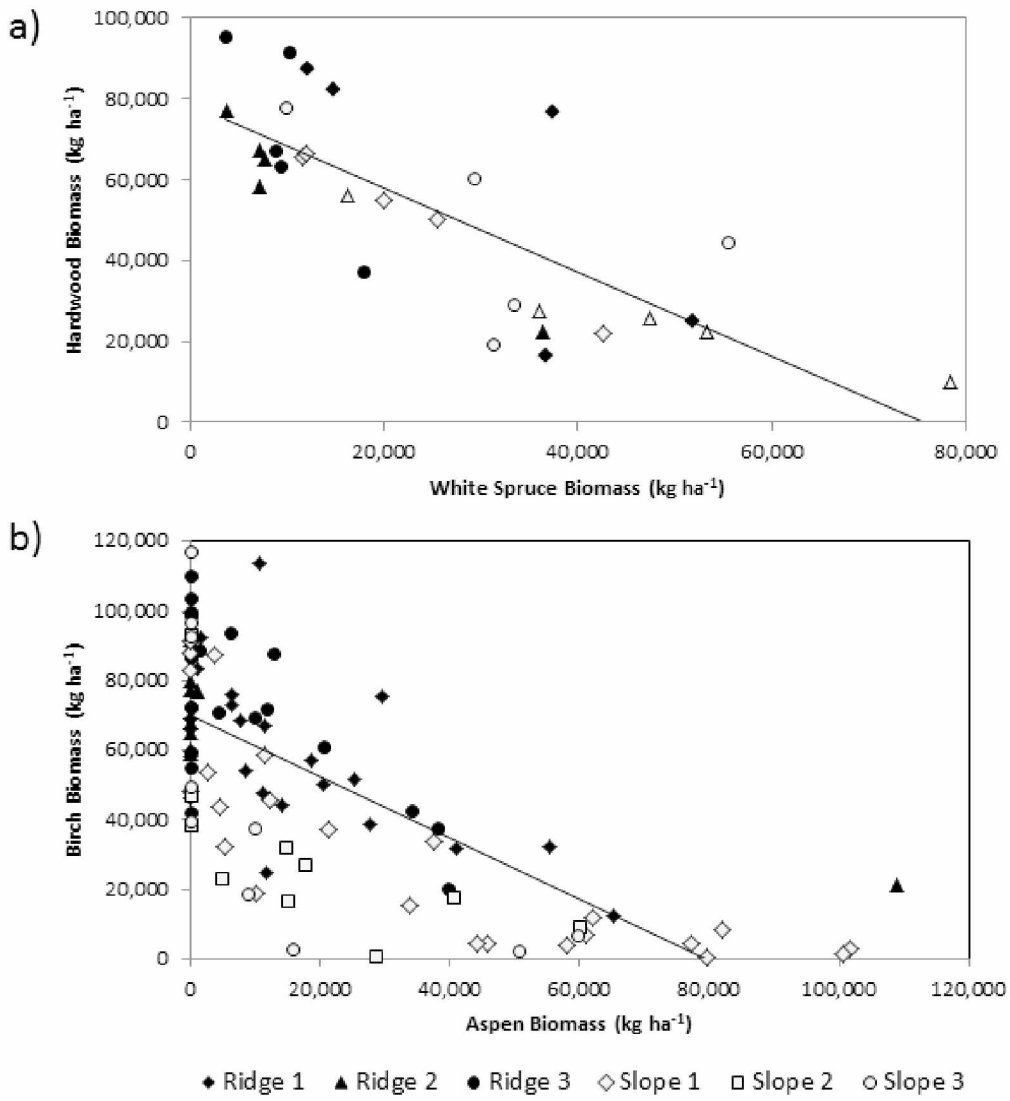

Figure 2.5: Biomass trade-offs between tree species, by block. a) Hardwood (aspen + birch) biomass as a linear function of white spruce biomass for 30 planted seedling S/SPs (Y = 78761 – 1.041 * X; R^2 = 0.61, p<0.0001). Planted seedlings in R2 and R3 were stunted due to infection with a greenhouse fungus. b) Aspen biomass as a linear function of birch biomass for 105 unplanted S/SPs (Y = 70,014 – 0.88 * X; R^2 = 0.52, p<0.0001).

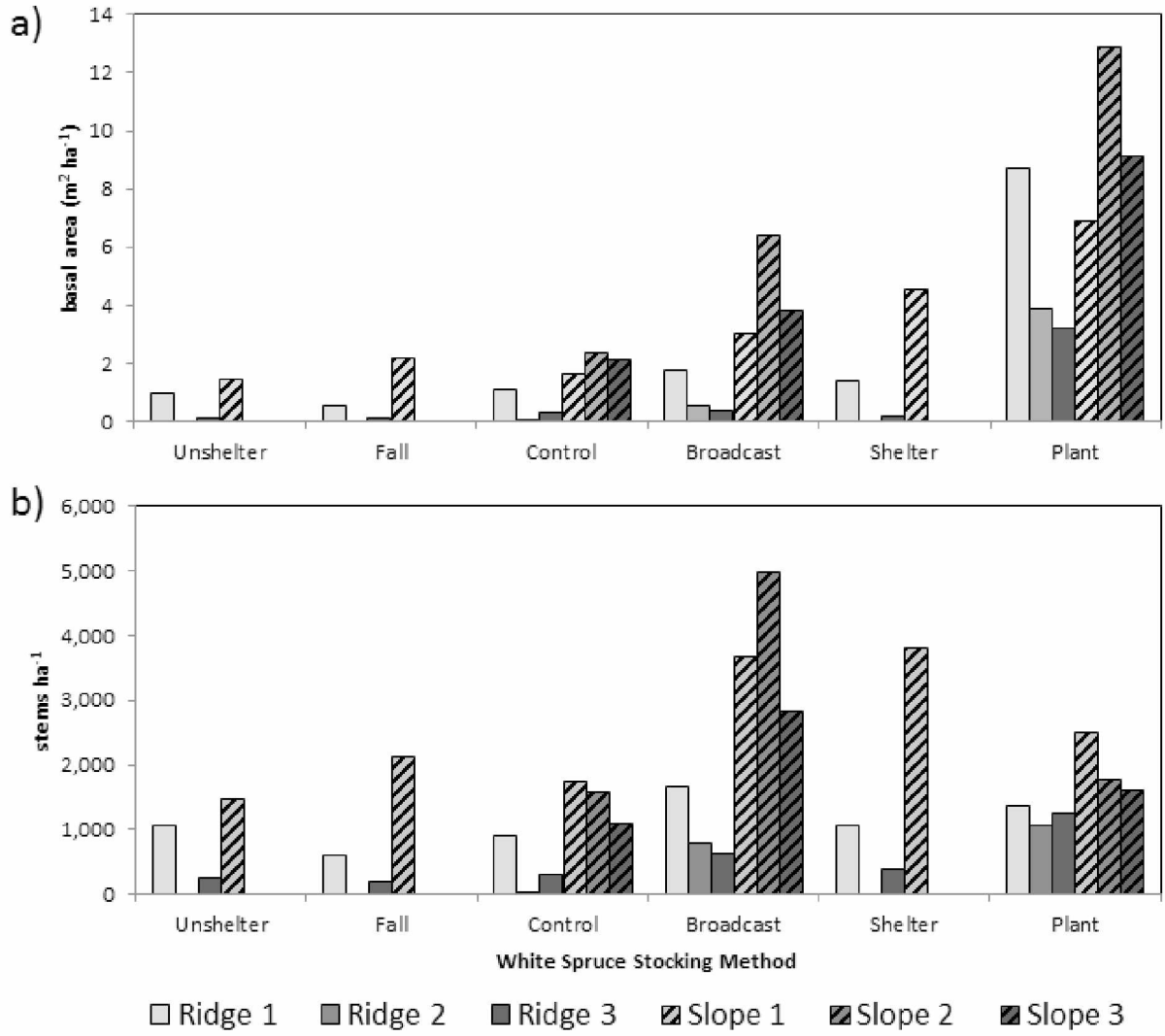

Figure 2.6: White spruce basal area (a) and stem density (b) across all 6 blocks and all 6 planting methods. Note that all blocks were sampled for Control (natural seedfall), Broadcast, and Plant (planted seedlings). Ridge Block 1, Ridge Block 3, and Slope Block 1 were sampled for Unshelter (spring unsheltered spot seeding), Fall (fall unsheltered spot seeding), and Shelter (spring sheltered spot seeding). N=5 for each unique combination of WSRT and Block.

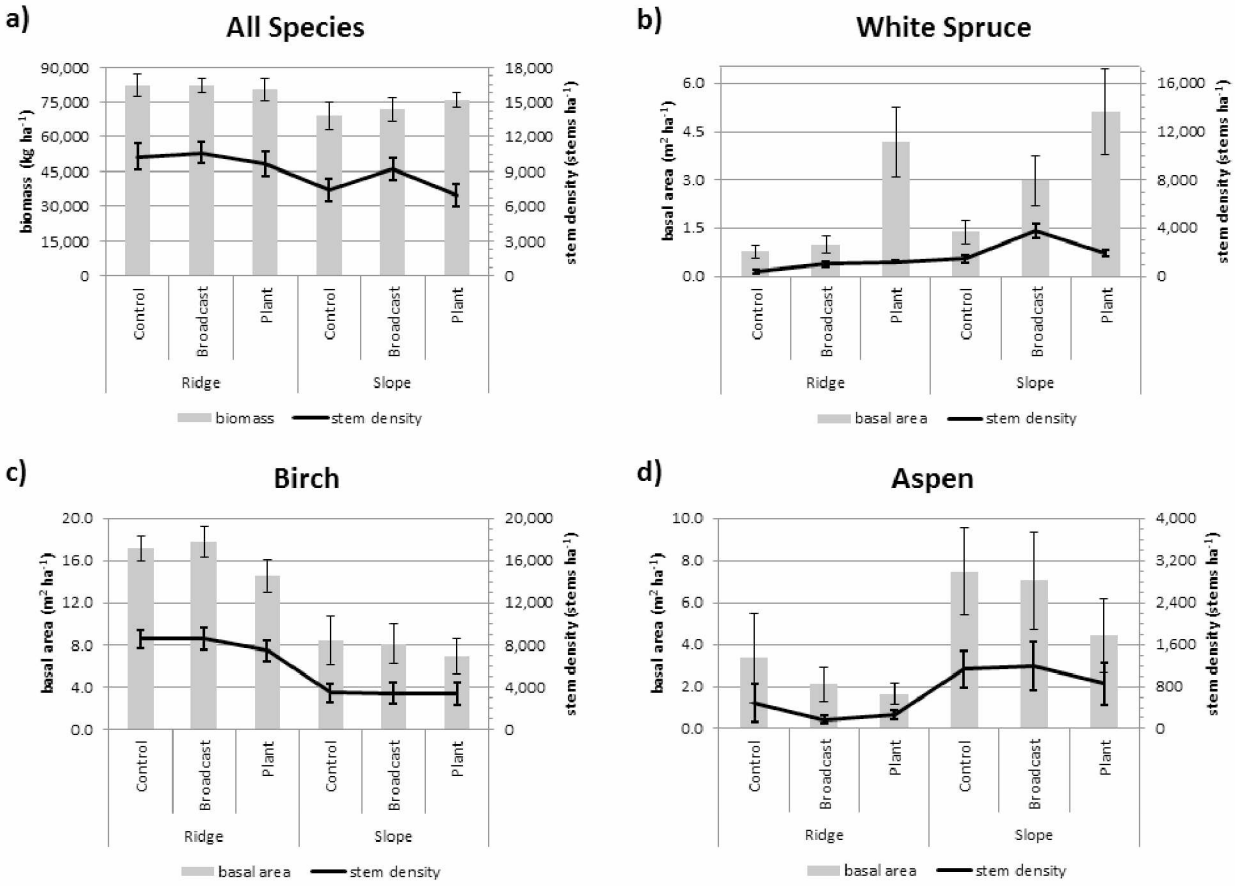

Figure 2.7: Plant by landform group means for all species (a), white spruce (b), birch (c), and aspen (d); figures from the 3WSRT dataset. Error bars are ± 1 SE.

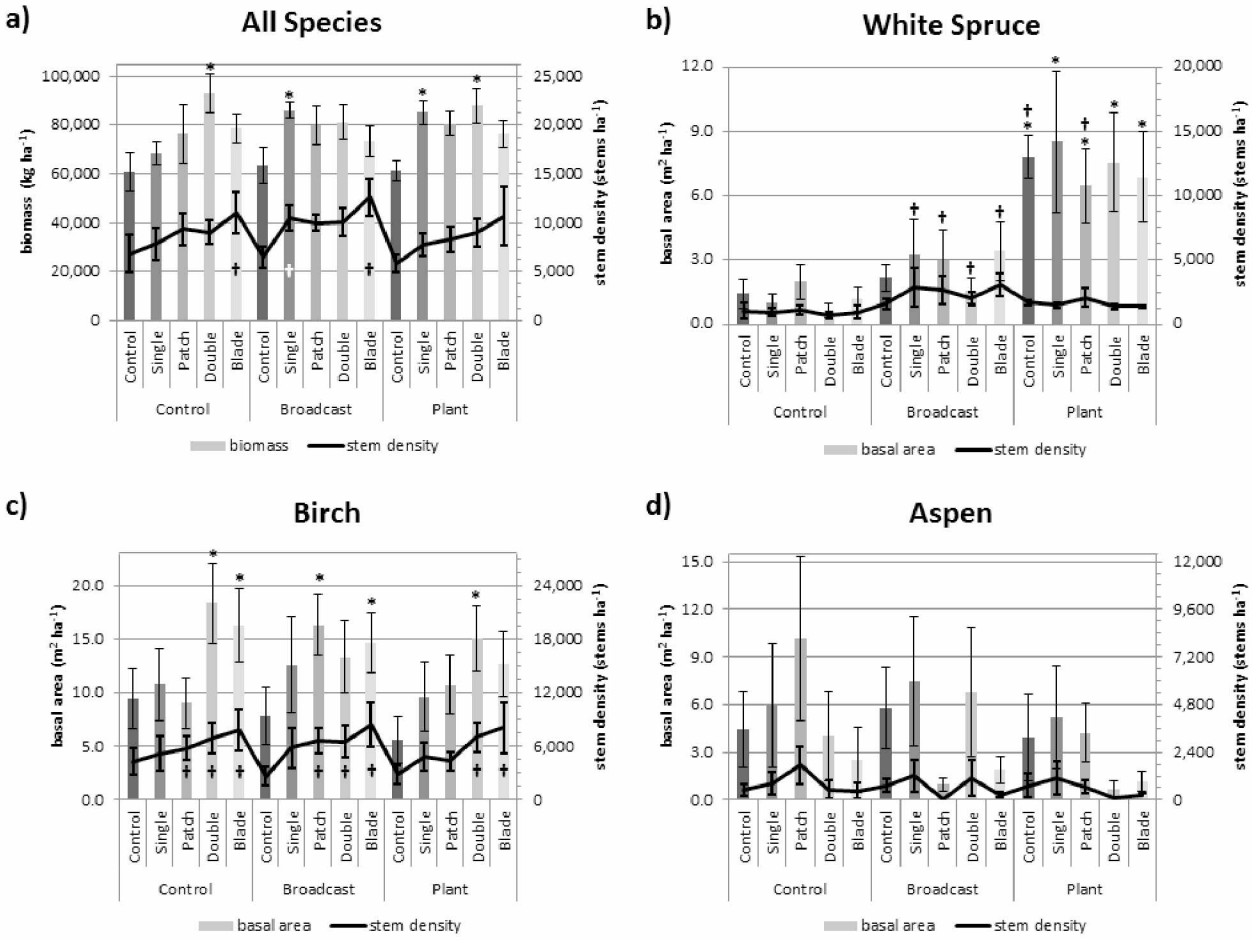

Figure 2.8: WSRT by GST group means for all species (a), white spruce (b), birch (c), and aspen (d); figures from the 3WSRT dataset. Error bars are ± 1 SE.

* Basal area significantly different from non-scarified, natural seedfall control using Dunnett ($p<0.05$)

† Stem density significantly different from non-scarified, natural seedfall control using Dunnett ($p<0.05$)

Table 2.8: ANOVA table for the 6WSRT dataset, measured on Ridge Block 1, Ridge Block 3, and Slope Block 1. GST = ground scarification treatment factor and WSRT = white spruce regeneration treatment. Specific linear contrasts are listed at bottom of each table for each response variable.

* significant result ($\alpha<0.05$)

		White Spruce			
		basal area		stem density	
	df	F-value	p-value	F-value	p-value
GST	4	0.52	0.7264	0.22	0.9200
WSRT	5	15.19	<0.0001*	7.28	<0.0001*
WSRT*GST	20	0.64	0.8545	1.04	0.4410
scar vs nonscar	1	2.56	0.1174	1.82	0.1842

Chapter 3 – General Conclusion: Management implications for Alaskan forest practices

3.1. Management in Alaska

The boreal forest is experiencing an accelerating wildfire disturbance regime (Kasischke and Turetsky, 2006; Calef *et al.*, 2015), as well as increasing utilization of both green and salvage-harvested timber resources (Nappi *et al.*, 2004). Given this backdrop of uncertainty and the marginal economics of investments in boreal Alaska forest management, forest managers require up-to-date scientific knowledge to identify the most effective practices and allocate scarce resources. The study discussed in the previous chapter highlights the continued effects of silvicultural practices undertaken early in a stand's post-fire development. We identified trade-offs in composition and dominance among major upland tree species as a consequence of operational practices targeting white spruce regeneration (Figure 3.1). Specifically, a better understanding of site preparation by mechanical ground scarification, assisted white spruce stocking, and topographical position allows for more effective management of productive upland forests. The results found in examining the stem exclusion stage of this large experimental trial have important implications for wood biomass silviculture, mixed species management, and landscape level structural diversity, but this study's domain of inference must be carefully interpreted.

We examined a single burned and salvage-harvested stand in a mixed spruce-hardwood forest in the south-facing uplands of Interior Alaska. Though our findings comprehensively characterize the interactions on this stand 30 years after fire, we caution against over-generalization to different site types or distant regions. Sites that were burned only or harvested only experience different recruitment conditions, specifically with regard to seed sources and seedbed conditions (discussed in Chapter 1). Also, this regeneration trial was established in a specific time period characterized by two large white spruce seed masting episodes and particular seasonal weather conditions. However, the results from this study are highly relevant to Alaskan forest practices compared to findings from more southerly areas of the boreal forest due to variation of environmental characteristics across latitude. For example, balsam poplar (*Populus balsamifera* L.) ecotypes are adapted to localized climate and photoperiod, and experience different growth patterns when planted outside of their native region (Robertson, 2012). Wildfire regimes vary across the Canadian boreal forest, and regional populations of boreal tree species adapt to localized selection pressures and recruitment conditions (Bergeron *et al.*, 2004). A meta-analysis of boreal forest mega-transects demonstrated substantial variation across latitude in climate,

vegetation types, vegetation-mediated climate interactions, and water/energy fluxes (McGuire *et al.*, 2002). In addition, the study area typifies productive upland ecosystems managed for timber production in the Tanana Valley State Forest in terms of climate, soils, and vegetative communities (Crimp *et al.*, 1997). Most forest harvest in the 5.8 million ha Tanana basin takes place on south-facing uplands or floodplains, typically in stands with a significant white spruce component (DOF, 2001; Wurtz *et al.*, 2006). Finally, we used measurements that are inexpensive to obtain, such as basal area and biomass estimates based on dbh only, which facilitates comparison of our study results with other regions. While the precise magnitude of a stand's response to regeneration treatments may vary somewhat across Interior Alaska, similar factors control forest growth throughout the region. For example, white spruce volume throughout Alaska's boreal forest can be predicted with 99% accuracy from a single allometric equation (Malone *et al.*, 2013). We believe the relationships and mechanisms reported have wide applicability throughout Interior Alaska's managed forest.

Alaskan forest managers are obliged to provide for multiple uses via the sustained yield principle (Alaska Constitution, Article VIII). Sustained yield mandates that present activities maintain a non-declining productive capacity of a renewable resource in perpetuity (Neumayer, 2003). According to AS (Alaska Statute) 41.17.200, "The primary purpose in the establishment of state forests is timber management that provides for the production, utilization, and replenishment of timber resources while allowing other beneficial uses of public land and resources." In order to maintain sustainability of the timber base, the annual allowable cut of state-designated forestry lands is calculated with the area control method based on stand types and rotation age estimates (DOF, 2001). Acreage disturbed by wildfire as well as green harvest both count toward the annual allowable cut (Douglas Hanson, personal communication). When a greater area is burned, harvest is shifted toward salvage operations. As a result, salvage harvest is an important practice for Alaska's timber industry historically, and it continues to be today (Crimp *et al.*, 1997; DOF, 2013; Hermanns, 2013). The need to regularly shift the type of harvest requires research that focuses on the unique silvicultural characteristics of burned and salvage-harvested sites.

Following harvest, Alaskan forest managers follow the Forest Resources and Practices Act (FRPA) to comply with sustained yield requirements (AS 41.17, implemented as 11 Alaska Administrative Code 95). FRPA requires preparation of a reforestation plan specifying target species composition and stocking levels prior to harvest, including any site preparation or assisted regeneration that may be necessary (DOF, 2007). If reforestation plans rely on natural regeneration, landowners must ensure the presence of suitable seed sources or clonal species. Appropriate silvicultural practices are essential: the

Tanana Valley State Forest management plan requires, "...tailoring reforestation techniques to ensure seedbed availability and adequate regeneration of a range of native upland forest types" (DOF, 2001). As this is being written, the Alaska Board of Forestry is reviewing reforestation standards and best management practices for Interior Alaska (DOF, 2014), and our study provides timely input for this process.

The decision to undertake a salvage harvest following wildfire requires careful consideration of economic and ecological factors (Figure 3.2). Ecological factors broadly include specific structural requirements for wildlife habitat (e.g., snags; see Nappi *et al.* (2004)) or administratively protected areas (e.g., riparian areas; see FRPA regulations in DOF (2007)). If any of these ecological values becomes and overriding concern on a particular burned site, managers may opt not to salvage harvest (or else do so in a way that mitigates the ecological concerns, e.g., retaining a buffer around riparian areas). Economic factors often are a major constraint in salvage operations. Economic factors include the cost versus benefit of harvesting on-site burned timber, and consideration of reforestation expenses that may increase the cost of management activities. If however, economic and ecological concerns can be addressed, a harvest prescription and reforestation plan can be developed for a burned site. Pre-planning is essential for time-sensitive salvage operations to be carried out in a thoughtful way (Nappi, 2011), as burned timber loses value rapidly but hasty salvage plans may not fully ameliorate ecological and economic concerns.

3.2. Wood Biomass in Alaska

Alaska has historically experienced high energy costs, but is in the midst of a socio-economic shift back toward wood energy. Many rural Alaskan communities rely on diesel oil for heat and electricity, and even in larger municipalities a substantial share of home heating comes from fuel oil (Nicholls *et al.*, 2010). High energy prices, however, are sustaining an increased demand for fuelwood for private residences as well as for public buildings in smaller communities (AEA, 2011). The Alaska Energy Authority (AEA) has conducted 56 wood energy feasibility studies since 2008, with 73% of those from 2012-2014. Alaska is a rural state with large forest acreage that could be harvested for wood biomass (Fresco and Chapin, 2009). Timber harvest has historically been small (Wurtz *et al.*, 2006), but the expanding use of wood biomass for heat generation is increasing demand for Interior Alaska's forest resource in terms of cordwood, biomass chips, and pelletized products (AEA, 2011). However, conventional silvicultural practices to produce saw logs and cordwood may not efficiently produce wood biomass. For example, in many types of biomass harvest, all of a tree may be utilized and conventionally undesirable tree species may be utilized (Manley and Richardson, 1995). Biomass dollar value is often

less per unit volume than conventional roundwood products, but total volume recoverable from a biomass-harvested stand may be as much as 50% higher with a shorter rotation length.

Appropriate silvicultural practices depend on the desired product, and management that generates large-dimension sawtimber from softwood species may not efficiently maximize total biomass. One of the first steps in a wood energy biomass management plan is to assess the supply of the local timber resource over time (Becker *et al.*, 2014). Supply can come from the existing stock of forested stands, or from short rotation plantations of fast-growing woody species. Short rotation biomass plantations require frequent harvests (i.e., every 2-8 years) and regular fertilization to replace depleted soil nutrients. Typical biomass crop species may be planted from cuttings, grow quickly, and resprout vigorously when coppiced. Willows (*Salix* spp.), alders (*Alnus* spp.), and poplars (*Populus* spp.) are considered potential candidates for this type of harvest in Interior Alaska (Garber-Slaght *et al.*, 2009). However, short rotation biomass in Alaska is limited by a brief growing season, highly variable summer precipitation, and low availability of soil nutrients, which lead to considerably lower per hectare yields of biomass compared to temperate regions (Byrd, 2013). Additionally, Interior Alaska's forest resource provides low value per acre even on productive sites, and capital investments like fertilization are often constrained (Wurtz *et al.*, 2006). Natural stands managed at minimal cost are available throughout the state, and biomass harvests near communities generate additional benefits by reducing wildfire risk (Nicholls *et al.*, 2010). Forest managers seeking wood biomass may opt to harvest existing growing stock instead of installing and maintaining intensive biomass plantations.

If management for biomass focuses on natural stands, particularly salvaging burned sites, rather than short rotation plantations, the ability to reforest under a constrained budget depends on the legacy effects following the fire (Figure 3.3). Legacy effects of particular importance for upland forests include the post-fire configuration of unburned stands near the salvaged stand, particularly the autumn prevailing wind direction, as well as the pre-fire basal area of asexual sprouting species (e.g., aspen). Scarification is not a blanket way to regenerate more biomass, and its effects will be mediated by these legacy conditions. For example, we found that intensive scarification methods like blade or double disc increased birch regeneration but decreased aspen. We also identified a trade-off between birch and aspen on our study area, indicating that the two species may segregate based on pre-fire basal area and incoming solar radiation.

The least cost management approach to reforest a site would involve neither scarification nor artificial stocking. On sites with a substantial pre-fire aspen component, doing nothing appears to maximize stand biomass (particularly avoiding intensive scarification methods such as bulldozer blading

or multiple passes with a disc-trencher). However, this management approach requires selecting sites that are conducive to aspen (Kurkowski *et al.*, 2008), particularly with substantial pre-disturbance root systems (Frey *et al.*, 2003). Typical practices to regenerate aspen in Interior Alaska were reviewed by Paragi and Haggstrom (2007), who found prescribed burns, chainsaw felling, and shearblading all produced large numbers of aspen stems (34,800 – 233,000 stems ha^{-1}). Prescribed fire was found to produce more variable stocking and required complete top-kill of aspen trees; any residual leaf area would suppress aspen suckering through the production of auxins (Paragi and Haggstrom, 2007). Warmer forest floors following the removal of logging slash was correlated with an increase in aspen sucker reproduction (Bella, 1986). High-severity wildfire similarly reduces ground-cooling organic layers, ground layer vegetation cover, and woody debris while killing off the aspen canopy (Ilisson and Chen, 2009). Aspen regenerated prolifically after the Rosie Creek Fire, with estimates between 71,600 – 138,150 stems ha^{-1} on pre-fire quaking aspen sites (Foote and Viereck, 1985; MacCracken and Viereck, 1990). The highly intensive scarification methods employed in this experiment reduced aspen stem density, and high-severity fire alone is likely sufficient to regenerate aspen.

By contrast, on sites with a large pre-fire birch component and nearby birch seed sources, scarification will increase total biomass (specifically birch biomass). Birch abundance in our study was related to distance to unburned edge (with autumn prevailing wind direction not significant), emphasizing the need for a proximate seed source before considering of scarification. Increasing scarification intensity, as measured by mineral soil exposed, likely increases birch on a site up to a critical point where overcrowding may occur. We found double-disc plowing that exposed 25% mineral soil increased birch basal area the most, an effect that may be related to the depth of scarification and the soil horizons exposed (Densmore and Page, 1992). Scarification for birch reproduction may also increase moose activity due to the increase of young trees (Cole *et al.*, 1999), and should therefore be undertaken with full knowledge of local wildlife dynamics. Experimental moose exclusion in Finland permitted increased survival and growth of both silver birch (*Betula pendula* Roth) and European aspen (*Populus tremula* L.) (den Herder *et al.*, 2009). Moose preferentially browse hardwood saplings such as birch, willow, and aspen, which in turn can affect successional trajectory toward greater conifer dominance (Kielland and Bryant, 1998).

Though planting white spruce seedlings had a neutral effect on total biomass in this study after 28 years, this practice may increase total woody biomass over a longer time period by creating a multi-species stand. Some boreal silviculture systems actively cultivate a spruce understory beneath a birch canopy, which protects young spruce from frost and herbivory as well as improves soil qualities (Simard

and Hannam, 2000; Wurtz and Zasada, 2001). Small spruce seedlings do not impact birch growth, and eventual removal of the birch overstory releases the shade-tolerant spruce (Simard and Hannam, 2000). White spruce and aspen mixed stands in Alberta also show higher gross productivity when considered for rotation ages over 80 years due to partitioning of site resources in space and time (Man and Lieffers, 1999). Our results indicate that planting white spruce seedlings shifts more of a site's productivity from hardwood to white spruce growth, but could be a viable strategy to produce more biomass over a long time period.

Forest managers considering biomass harvests must weigh the likelihood of depleting site productivity over multiple rotations. Site productivity is defined as the "ability of a given site to accumulate biomass over a given period of time," and is closely related with net primary productivity (Morris, 1997). As Grigal (2000) notes, "It is axiomatic that forest management activities alter soil physical, chemical, and biological properties." However, the duration, extent, and magnitude of those effects have diverse implications depending on the specific plant community, geography, and management activity in question. Following a wood biomass harvest, there may be reductions in soil organic matter, increased soil warmth, decreased nutrient pools due to leaching and material removal, pH changes, and higher levels of soil biota activity (e.g., microbe community, mycorrhizae) among other effects (Grigal, 2000; Lattimore *et al.*, 2009). Biomass harvest typically removes the whole tree, including twigs and foliage that hold a high proportion of site nutrients (Manley and Richardson, 1995). For example, whole tree harvest in Sweden reduced exchangeable cations of K, Ca, Mg, Mn, and Zn and increased acidity compared to stem only harvests (Olsson *et al.*, 1996). For analogous wildfire disturbance, effects on biogeochemistry are quite variable depending on severity and fire duration (Kimmins, 2004). For example, post-fire decreases in available N and increases in P have been documented, but changes are generally neutral in the long-term (Certini, 2005).

A program to develop best management practices through long-term site monitoring, field experimental trials, and computer-based modeling of multiple rotations has been recommended as a way to conserve site quality in Canada (Morris, 1997). A set of criteria, forest health indicators, and verification checklists have been developed to plan for, implement, and subsequently monitor the sustainability of wood energy harvests (Lattimore *et al.*, 2009). A site's soil and nutrient capital is a resource that renews very slowly, and long-term productivity after repeated biomass harvests may decline without careful husbandry of soil and nutrient resources.

3.3. Large Dimension White Spruce

White spruce sawtimber has been the primary target for commercial harvests in Interior Alaska in recent decades (Wurtz *et al.*, 2006), and regenerating post-fire salvaged stands to establish future white spruce crop trees is an important objective for many land managers. However, changing climatic conditions, particularly warmer summers and less predictable rainfall, increase the challenges to foresters trying to maintain white spruce forest types on the landscape (Chapin *et al.*, 2010). Landscape-scale models suggest a shift from conifer to hardwood-dominated forests due to forecasted changes in fire regime and climate (Mann *et al.*, 2012), especially ecosystems with white spruce (Juday *et al.*, 2003). Dendrochronological research across Alaska, Yukon Territory, and British Columbia found a negative growth response of white spruce to increased temperatures for stands growing on a warmer, drier sites already near white spruce's lower moisture thresholds (Barber *et al.*, 2000; Miyamoto *et al.*, 2010). There is some indication that upland white spruce sites derive much of their growing season soil moisture from snowmelt, and may be more resilient to reduced summer precipitation compared to floodplain white spruce sites (Yarie, 2008). Efforts to maintain existing forest types will likely focus on maintaining white spruce on the landscape, and upland white spruce stands may be a well-buffered forest type to climate change. However, pure stands of white spruce are relatively uncommon on the landscape (Crimp *et al.*, 1997; Hanson, 2013), and spruce typically occur in mixed species stands associated with birch or aspen until older seral stages are reached after approximately 100 years (Chapin *et al.*, 2006). Therefore, the development of effective management strategies for all major upland species is crucial to sustaining the forest resource.

Regenerating large dimension white spruce following fire and salvage harvest in the Interior Alaska uplands requires both establishing vigorous spruce and suppressing other competitors from the earliest years of stand establishment (Figure 3.4). Among the practices evaluated in this study, the single most effective way to increase white spruce basal area and biomass was to plant white spruce seedlings. Planted seedlings succeeded on sites even where bluejoint grass (*Calamagrostis canadensis*) likely suppressed direct seeding efforts. Youngblood and Zasada (1991) speculated that planted white spruce seedlings experienced reliably high survival on their floodplain study site due to the nature of seasonally deposited soils. Our finding of reliable survival and growth across both slope and ridge landforms indicates that planted seedling success may be widespread across multiple Interior Alaska topographies.

Some studies have suggested scarification as an effective way to reduce grass competition and enhance spruce establishment (Collins and Schwartz, 1998), but the scarification methods employed in the RCFTRI were not intensive enough to affect sites with substantial pre-existing cover. Fire-thinned

forest floor organic layers likely are sufficiently suitable seedbeds to establish white spruce from seed, provided grass cover is low to minimal. Where grass cover is vigorous and dense, for example on the ridge site, even scarification likely will not improve spruce establishment due to rapid grass regrowth. Broadcast seeding, though perhaps less investment than planted seedlings, resulted in many smaller stems that will have delayed entry into the canopy (and thus longer rotation lengths) if they are able to at all.

We did not find that the scarification methods employed in this study effectively increased white spruce basal area, and in fact double-disc scarification tended to reduce white spruce basal area from natural or direct seeding between 29-44%. A British Columbia study found that patch scarification stimulated an increase in vegetative competition and reduced spruce seedling growth (Bedford *et al.*, 2000). We hypothesize a similar competitive mechanism reduced growth where white spruce germinated in situ on double-disc scarified S/SPs, rather than having a two year head-start over competing vegetation as planted seedlings did.

The least-cost method to recruit spruce stems is from natural seedfall, but regeneration from natural seedfall requires both a significant seed source input and suitable seedbeds for germination (Alden, 1985; Rupp, 1997). The study area had both factors, and we found substantial white spruce recruitment throughout RCFTRI at distances up to 265 m downwind from unburned stands with a mature spruce component. However, several factors likely elevated the natural recruitment above what could be reliably expected in other post-fire salvage situations. For example, this study area experienced multiple masting episodes within a few years of disturbance (including the 1987 record high seedfall in 60 years of measurement), all spruce seed sources were uphill at higher elevation than the study area (which may increase horizontal distance before a seed reaches the ground), and even mortally injured trees contributed seed (Zasada, 1985). Other literature suggests approximately 100 m as a reliable downwind dispersal distance (Dobbs, 1976; Zasada, 1985; Greene *et al.*, 1999), and the unique factors of this study area suggest managers not depend on the long dispersal distances we found.

In addition to establishing white spruce, management for large dimension white spruce must also reduce the effects of hardwood competition on white spruce growth in order to minimize rotation lengths. Unfavorable light environments for understory conifer growth in mixed conifer-broadleaf stands are typically associated with slow growth and delayed canopy entry (MacIsaac and Navratil, 1995). Light transmittance through birch canopy is inversely related to birch basal area (Comeau *et al.*, 1998), and reduced light transmittance is associated with reduced white spruce basal area (Comeau *et al.*, 2003). Stands with a substantial regenerating birch component, especially stands in which the initial

accumulation of birch basal area was accelerated by early scarification treatments, will almost certainly support lower levels of white spruce basal area (originated from seed) due to the early birch canopy dominance. Similarly, aspen overstories that reduce light transmittance below 60% decrease white spruce growth, but levels above 40% allow for vigorous understory grass and shrub growth that may compete for other growth factors (Wright *et al.*, 1998). Management of light transmittance is an important consideration for mixed boreal stands (Simard and Hannam, 2000; Comeau *et al.*, 2005).

3.4. Achieving landscape-scale objectives

Salvage harvest and assisted regeneration may address some of the issues expected in forest ecosystems facing increasing fire regimes and climate change dislocations, but only partially address concerns about ecosystem integrity and diversity that typically are prime goals of ecologically-based forestry management. These practices combined may reduce harvest of unburned (green) stands, perpetuate timber production and associated forest economies, facilitate reforestation through assisted regeneration in severely burned areas, and permit more landscape level stand age diversity (see Chapter 1). For example, much of Interior Alaska's productive timber stock is middle-aged between 70-110 years (Hanson, 2013), with fewer young stands due to extensive timber harvest during the Gold Rush period of the early 20th century and successful wildfire suppression (Crimp *et al.*, 1997; Wurtz *et al.*, 2006). The lack of stand age diversity points to an emerging future challenge in supplying a continuous sustained yield of timber as well as young stand types that game species depend upon (DOF, 2001). The practice of salvage harvest can permit unharvested stands to reach older growth stages, while assisted regeneration ensures reproductive success in younger stages. However, salvage harvest introduces an unnatural second disturbance with wide-ranging effects on species diversity, stand composition and structure, and soil and nutrient cycling processes; assisted regeneration only directly ameliorates the stand composition effects of salvage. Large scale salvage harvest will likely require intensive silvicultural practices to ensure the ecological integrity and sustainability of the forest resource.

High-intensity management aims to maximize stand productivity through the application of silvicultural practices. When the timber resource has high value and harvest intensity is high, significant investment in practices such as assisted regeneration and thinning shorten the rotation length while producing a carefully regulated stand age structure that provides for a sustained timber yield (Seymour and Hunter, 1999). High intensity management, like salvage harvest and assisted regeneration, is a useful technique to improve forest and community resilience where the expanding fire footprint or climate-induced stress render regeneration success low or uncertain (Ogden and Innes, 2007). These

practices may become more expensive to continue in the face of increasing climate change, as current phenotypes become ill-adapted to the local climate and disturbance regime (Millar *et al.*, 2007). Also, boreal forest old growth stands characterized by understory re-initiation, gap-phase dynamics, and a long time since disturbance have received little attention in the past. Research suggests more frequent wildfires reduce the old growth component of a landscape, but even in the high frequency fire regime of the western Canadian boreal forest old growth maintains an important ecological footprint (Kneeshaw and Gauthier, 2003). High intensity management de-emphasizes stand age and species diversity in favor of fiber production and a carefully regulated age structure, which may in turn reduce the resilience of the forest to interacting disturbances.

Zonation is a landscape-scale strategy that divides the land base among management regimes with different goals and intensities. High intensity management practices (e.g., salvage harvest and assisted regeneration) are focused on part of a landscape, and on other areas managers may reduce human impact and facilitate natural development. For example, Alaska is currently zoned into different management intensities with regard to wildfire, with only 17% of the state's land area receiving active suppression (DeWilde and Chapin, 2006). Similarly for timber products, Seymour and Hunter (1999) propose high intensity management in some areas, which allows other areas to incorporate key components of forest health and resilience (e.g., older stand ages, more diverse species composition, and heterogeneous physical structure). The practice of post-fire salvage further enables a shift of harvest effort from live (green) stands toward burned stands, which reduces the harvest of green stands to meet wood fiber objectives. Unharvested green stands may thus increase in age, promoting stand age-diversity and its biodiversity correlates (Kneeshaw and Gauthier, 2003). At the same time, assisted regeneration following salvage can quickly restore forest cover in a burned stand. Expedient re-establishment of forest cover sustains the provision of ecosystem services to society, and minimizes the need of future generations to look toward reserved areas to meet their demands. However, if post-fire salvage harvests are widespread, the usually heterogeneous disturbance mosaic becomes more uniform both in terms of post-fire seedbeds and residual stand structure (Nappi, 2011). Application of salvage harvests and intensive regeneration techniques following every wildfire may reduce landscape-scale biodiversity and natural disturbance mosaics (Lindenmayer *et al.*, 2008).

Rather than recommending the automatic application of the silvicultural practices evaluated in the RCFTRI experiment to each burned stand, it is critically important to consider a management prescription in the context of the entire landscape. The "Triad approach" has been recommended in fire-prone areas of Canada's boreal forest, which envisions a portion of the landscape set aside for high-

intensity management for fiber production, some areas for strict conservation, and an intermediate regime permitting multiple uses managed within ecological forestry principles (Klenk *et al.*, 2009; Saint-Germain and Greene, 2009). Wildfire alone reduces the ecosystem services provided in the short term from a particular stand, but builds resilience into the larger spatial scales that sustain a higher overall level of ecosystem values to society (Nappi, 2011). Management practices that focus on ameliorating the landscape-scale implications of green and salvage harvest may buffer the landscape from substantial loss in ecosystem service value. The practices evaluated in this study provide valuable tools to managers concerned with such landscape level productivity, diversity, and long-term ecosystem sustainability in light of a changing climate.

3.5. References

AEA, 2011. Renewable energy atlas of Alaska. Alaska Energy Authority

Alden, J., 1985. Early surival and growth of white spruce on natural sites, in: (Eds.) Dyrness, C.T., Juday, G.P., Early Results of the Rosie Creek Fire Research Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK, pp. 40-43.

Barber, V.A., Juday, G.P., Finney, B.P., 2000. Reduced growth of Alaskan white spruce in the twentieth century from temperature-induced drought stress. Nature 405, 668-673. doi: 10.1038/35015049.

Becker, D., Lowell, E., Bihn, D., Anderson, R., Taff, S., 2014. Community Biomass Handbook Volume I: Thermal Wood Energy. U.S. Department of Agriculture, Forest Service PNW-GTR-899.

Bedford, L., Sutton, R.F., Stordeur, L., Grismer, M., 2000. Establishing white spruce in the Boreal White and Black Spruce Zone. New Forests 20, 213-233. doi: 10.1023/A:1006774518199.

Bella, I.E., 1986. Logging practices and subsequent development of aspen stands in east-central Saskatchewan. Forestry Chron. 62, 81-83. doi: 10.5558/tfc62081-2.

Bergeron, Y., Flannigan, M., Gauthier, S., Leduc, A., Lefort, P., 2004. Past, current and future fire frequency in the Canadian boreal forest: implications for sustainable forest management. Ambio 33, 356-360. doi: 10.1579/0044-7447-33.6.356.

Byrd, A., 2013. Evaluating short rotation poplar biomass on an experimental land-fill cap near Anchorage, Alaska. In, Department of Civil and Environmental Engineering. University of Alaska, Fairbanks, AK, p. 49.

Calef, M., Varvak, A., McGuire, A., Chapin, F., Reinhold, K., 2015. Recent changes in annual area burned in Interior Alaska: The impact of fire management. Earth Interactions In press. doi: 10.1175/EI-D-14-0025.

Certini, G., 2005. Effects of fire on properties of forest soils: a review. Oecologia 143, 1-10. doi: 10.1007/s00442-004-1788-8.

Chapin, F.S., McGuire, A.D., Ruess, R.W., Hollingsworth, T.N., Mack, M.C., Johnstone, J.F., Kasischke, E.S., Euskirchen, E.S., Jones, J.B., Jorgenson, M.T., Kielland, K., Kofinas, G.P., Turetsky, M.R., Yarie, J., Lloyd, A.H., Taylor, D.L., 2010. Resilience of Alaska's boreal forest to climatic change. Can. J. For. Res. 40, 1360-1370. doi: 10.1139/X10-074.

Chapin, F.S., Viereck, L.A., Adams, P.C., Van Cleve, K., Fastie, C.L., Ott, R.A., Mann, D., Johnstone, J.F., 2006. Successional processes in the Alaskan boreal forest, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 100-119.

Cole, E.C., Newton, M., Youngblood, A., 1999. Regenerating white spruce, paper birch, and willow in south-central Alaska. Can. J. For. Res. 29, 993-1001. doi: 10.1139/x99-030.

Collins, W.B., Schwartz, C.C., 1998. Logging in Alaska's boreal forest: creation of grasslands or enhancement of moose habitat. Alces 34, 355-374. doi: N/A.

Comeau, P.G., Gendron, F., Letchford, T., 1998. A comparison of several methods for estimating light under a paper birch mixedwood stand. Can. J. For. Res. 28, 1843-1850. doi: 10.1139/x98-159.

Comeau, P.G., Kabzems, R., McClarnon, J., Heineman, J.L., 2005. Implications of selected approaches for regenerating and managing western boreal mixedwoods. Forestry Chron. 81, 559-574. doi: 10.5558/tfc81559-4.

Comeau, P.G., Wang, J.R., Letchford, T., 2003. Influences of paper birch competition on growth of understory white spruce and subalpine fir following spacing. Can. J. For. Res. 33, 1962-1973. doi: 10.1139/x03-117.

Crimp, P.M., Phillips, S.J., Worum, G.T., 1997. Resources on State Forestry Lands in the Tanana Valley. Division of Forestry, Alaska Department of Natural Resources

den Herder, M., Kouki, J., Ruusila, V., 2009. The effects of timber harvest, forest fire, and herbivores on regeneration of deciduous trees in boreal pine-dominated forests. Can. J. For. Res. 39, 712-722. doi: 10.1139/X08-208.

Densmore, R.V., Page, J.C., 1992. Paper birch regeneration on scarified logged areas in Southcentral Alaska. North. J. Appl. Forestry 9, 63-66. doi: N/A.

DeWilde, L., Chapin, F.S., 2006. Human impacts on the fire regime of Interior Alaska: interactions among fuels, ignition sources, and fire suppression. Ecosystems 9, 1342-1353. doi: 10.1007/s10021-006-0095-0.

Dobbs, R.C., 1976. White spruce seed dispersal in central British Columbia. Forestry Chron. 52, 225-228. doi: 10.5558/tfc52225-5.

DOF, 2001. Tanana Valley State Forest Management Plan: 2001 Update. Division of Forestry, Alaska Department of Natural Resources

DOF, 2007. Alaska Forest Resources & Practices Regulations. In. Division of Forestry, Alaska Department of Natural Resources, http://forestry.alaska.gov/pdfs/2009AFResourcesPracticesRegulations_June2007.pdf, p. 68.

DOF, 2013. Final decision best interest finding for Mississippi Fire salvage. Division of Forestry, Alaska Department of Natural Resources

DOF, 2014. Final Minutes, Region II-III Reforestation Science & Technical Committee, Meeting #1 – April 29, 2014. In: Division of Forestry, A.D.o.N.R. (Ed.), Fairbanks, AK, p. 8.

Foote, J., Viereck, L., 1985. Burn severity: its impact on the natural revegetation process following the Rosie Creek Fire, in: (Eds.) Dyrness, C.T., Juday, G.P., Early Results of the Rosie Creek Fire Research Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK, pp. 26-29.

Fresco, N., Chapin, F.S., 2009. Assessing the potential for conversion to biomass fuels in interior Alaska. U.S. Department of Agriculture, Forest Service PNW-RP-579.

Frey, B.R., Lieffers, V.J., Landhäusser, S.M., Comeau, P.G., Greenway, K.J., 2003. An analysis of sucker regeneration of trembling aspen. Can. J. For. Res. 33, 1169-1179. doi: 10.1139/x03-053.

Garber-Slaght, R., Sparrow, S.D., Masiak, D.T., Holdmann, G., 2009. Opportunities for woody biomass fuel crops in Interior Alaska. Agricultural and Forestry Experiment Station, School of Agriculture and Land Resources Management, University of Alaska Fairbanks

Greene, D.F., Zasada, J.C., Sirois, L., Kneeshaw, D., Morin, H., I. Charron, Simard, M.-J., 1999. A review of the regeneration dynamics of North American boreal forest tree species. Can. J. For. Res. 29, 824-839. doi: 10.1139/x98-112.

Grigal, D.F., 2000. Effects of extensive forest management on soil productivity. For. Ecol. Manage. 138, 167-185. doi: 10.1016/S0378-1127(00)00395-9.

Hanson, D., 2013. Timber inventory of state forest lands in the Tanana Valley 2013. Division of Forestry, Alaska Department of Natural Resources

Hermanns, J., 2013. Five year schedule of timber sales 2013-2017. Division of Forestry, Alaska Department of Natural Resources

Ilisson, T., Chen, H.Y.H., 2009. Response of six boreal trees species to standreplacing fire and clearcutting. Ecosystems 12, 820-829. doi: 10.1007/s10021-009-9259-z.

Juday, G.P., Barber, V., Rupp, T.S., Zasada, J.C., Wilmking, M., 2003. A 200-year perspective of climate variability and the response of white spruce in Interior Alaska, in: (Eds.) Greenland, D., Goodin, D.G., Smith, R.C., Climate variability and ecosystem response at long-term ecological research sites. Oxford University Press, New York, NY, pp. 226-250.

Kasischke, E.S., Turetsky, M.R., 2006. Recent changes in the fire regime across the North American boreal region: spatial and temporal patterns of burning across Canada and Alaska. Geophysical Res. Lett. 33, 1-5. doi: 10.1029/2006GL025677.

Kielland, K., Bryant, J.P., 1998. Moose herbivory in taiga: effects on biogeochemistry and vegetation dynamics in primary succession. Oikos 82, 377-383. doi: N/A.

Kimmins, J.P., 2004. Forest ecology: a foundation for sustainable forest management and environmental ethics. Macmillan Publishing Company, Upper Saddle River, NJ.

Klenk, N.L., Bull, G.Q., MacLellan, J.I., 2009. The “emulation of natural disturbance” (END) management approach in Canadian forestry: A critical evaluation. Forestry Chron. 85, 440-445. doi: 10.5558/tfc85440-3.

Kneeshaw, D., Gauthier, S., 2003. Old growth in the boreal forest: a dynamic perspective at the stand and landscape level. Environ. Rev. 11, 99-114. doi: 10.1139/a03-010.

Kurkowski, T.A., Mann, D.H., Rupp, T.S., Verbyla, D.L., 2008. Relative importance of different secondary successional pathways in an Alaskan boreal forest. Can. J. For. Res. 38, 1911-1923. doi: 10.1139/X08-039.

Lattimore, B., Smith, C.T., Titus, B.D., Stupak, I., Egnell, G., 2009. Environmental factors in woodfuel production: Opportunities, risks, and criteria and indicators for sustainable practices. Biomass and Bioenergy 33, 1321-1342. doi: 10.1016/j.biombioe.2009.06.005.

Lindenmayer, D.B., Burton, P.J., Franklin, J.F., 2008. Salvage Logging and its Ecological Consequences. Island Press, Washington, DC.

MacCracken, J.G., Viereck, L.A., 1990. Browse regrowth and use by moose after fire in Interior Alaska. Northwest Sci. 64, 11-18. doi: N/A.

MacIsaac, D.A., Navratil, S., 1995. Competition dynamics in juvenile boreal hardwood-conifer mixtures. In: Comeau, P.G., Thomas, K.D. (Eds.), Silviculture of temperate and boreal broadleaf-conifer mixtures. BC Ministry of Forests, Richmond, BC, pp. 23-34.

Malone, T., Liang, J., Packee, E.C., 2013. Total and merchantable volume of white spruce in Alaska. West. J. Appl. Forestry 28, 71-77. doi: 10.5849/wjaf.12-003.

Man, R., Lieffers, V.J., 1999. Are mixtures of aspen and white spruce more productive than single species stands? Forestry Chron. 75, 505-513. doi: 10.5558/tfc75505-3.

Manley, A., Richardson, J., 1995. Silviculture and economic benefits of producing wood energy from conventional forestry systems and measures to mitigate negative impacts. Biomass and Bioenergy 9, 89-105. doi: 10.1016/0961-9534(95)00082-8.

Mann, D.H., Scott Rupp, T., Olson, M.A., Duffy, P.A., 2012. Is Alaska's boreal forest now crossing a major ecological threshold? Arct., Antarctic, Alp. Res. 44, 319-331. doi: 10.1657/1938-4246-44.3.319.

McGuire, A.D., Wirth, C., Apps, M., Beringer, J., Clein, J., Epstein, H., Kicklighter, D.W., Bhatti, J., Chapin, F.S., de Groot, B., Efremov, D., Eugster, W., Fukuda, M., Gower, T., Hinzman, L., Huntley, B., Jia, G.J., Kasischke, E., Melillo, J., Romanovsky, V., Shvidenko, A., Vaganov, E., Walker, D., 2002. Environmental variation, vegetation distribution, carbon dynamics and water/energy exchange at high latitudes. Veg. Sci. 13, 301-314. doi: 10.1111/j.1654-1103.2002.tb02055.x.

Millar, C.I., Stephenson, N.L., Stephens, S.L., 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecol. Applications 17, 2145-2151. doi: 10.1890/06-1715.1.

Miyamoto, Y., Griesbauer, H.P., Green, D.S., 2010. Growth responses of three coexisting conifer species to climate across wide geographic and climate ranges in Yukon and British Columbia. For. Ecol. Manage. 259, 514-523. doi: 10.1016/j.foreco.2009.11.008.

Morris, D.M., 1997. The role of long-term site productivity in maintaining healthy ecosystems: A prerequisite of ecosystem management Forestry Chron. 73, 731-740. doi: 10.5558/tfc73731-6.

Nappi, A., 2011. Harvesting in burned forests — issues and orientations for ecosystem-based management. Ministère des Ressources naturelles et de la Faune, Direction de l'environnement et de la protection des forêts

Nappi, A., Drapeau, P., Savard, J.-P.L., 2004. Salvage logging after wildfire in the boreal forest: Is it becoming a hot issue for wildlife? Forestry Chron. 80, 67-74. doi: 10.5558/tfc80067-1.

Neumayer, E., 2003. Weak versus strong sustainability: exploring the limits of two opposing paradigms. Edward Elgar Publishing, Northampton, MA.

Nicholls, D.L., Brackley, A.M., Barber, V., 2010. Wood energy for residential heating in Alaska: current conditions, attitudes, and expected use. U.S. Department of Agriculture, Forest Service PNW-GTR-826.

Ogden, A.E., Innes, J., 2007. Incorporating climate change adaptation considerations into forest management planning in the boreal forest. Int. Forestry Rev. 9, 713-733. doi: 10.1505/ifor.9.3.713.

Olsson, B.A., Bengtsson, J., Lundkvist, H., 1996. Effects of different forest harvest intensities on the pools of exchangeable cations in coniferous forest soils. For. Ecol. Manage. 84, 135-147. doi: 10.1016/0378-1127(96)03730-9.

Paragi, T.F., Haggstrom, D.A., 2007. Short-term responses of aspen to fire and mechanical treatments in Interior Alaska. North. J. Appl. Forestry 24, 153-157. doi: N/A.

Robertson, A.L., 2012. Acclimation and migration potential of a boreal forest tree, balsam poplar (Populus balsamifera L.) in a changing climate. In, Department of Biology and Wildlife. University of Alaska, Fairbanks, AK, p. 162.

Rupp, T.S., 1997. A geographic model of landscape-level post-disturbance forest establishment patterns of Interior Alaska white spruce ecosystems. In, School of Natural Resources. University of Alaska, Fairbanks.

Saint-Germain, M., Greene, D.F., 2009. Salvage logging in the boreal and cordilleran forests of Canada: Integrating industrial and ecological concerns in management plans. Forestry Chron. 85, 120-134. doi: 10.5558/tfc85120-1.

Seymour, R.S., Hunter, M.L., Jr., 1999. Principles of ecological forestry, in: (Ed.) Hunter, M.L., Jr., Maintaining biodiversity in forested ecosystems. Cambridge University Press, Cambridge UK, pp. 22-61.

Simard, S.W., Hannam, K.D., 2000. Effects of thinning overstory paper birch on survival and growth of interior spruce in British Columbia: implications for reforestation policy and biodiversity. For. Ecol. Manage. 129, 237-251. doi: 10.1016/S0378-1127(99)00169-3.

Wright, E.F., Coates, K.D., Canham, C.D., Bartemucci, P., 1998. Species variability in growth response to light across climatic regions in northwestern British Columbia. Can. J. For. Res. 28, 871-886. doi: 10.1139/x98-055.

Wurtz, T.L., Ott, R.A., Maisch, J.C., 2006. Timber harvest in Interior Alaska, in: (Eds.) Chapin, F.S., Oswood, M.W., Van Cleve, K., Viereck, L.A., Verbyla, D.L., Alaska's Changing Boreal Forest. Oxford University Press, New York, pp. 302-308.

Wurtz, T.L., Zasada, J.C., 2001. An alternative to clear-cutting in the boreal forest of Alaska: a 27-year study of regeneration after shelterwood harvesting. Can. J. For. Res. 31, 999-1011. doi: 10.1139/cjfr-31-6-999.

Yarie, J., 2008. Effects of moisture limitation on tree growth in upland and floodplain forest ecosystems in interior Alaska. For. Ecol. Manage. 256, 1055-1063. doi: 10.1016/j.foreco.2008.06.022.

Youngblood, A., Zasada, J.C., 1991. White spruce artificial regeneration options on river floodplains in interior Alaska. Can. J. For. Res. 21, 423-433. doi: 10.1139/x91-057.

Zasada, J.C., 1985. Production, dispersal, and germination of white spruce and paper birch and first year seedling establishment after the Rosie Creek Fire, in: (Eds.) Juday, G.P., Dyrness, C.T., Early Results of the Rosie Creek Fire Research Project-1984. University of Alaska, Agricultural and Forestry Experiment Station, Misc. Publ. 85-2, Fairbanks, AK, pp. 34-37.

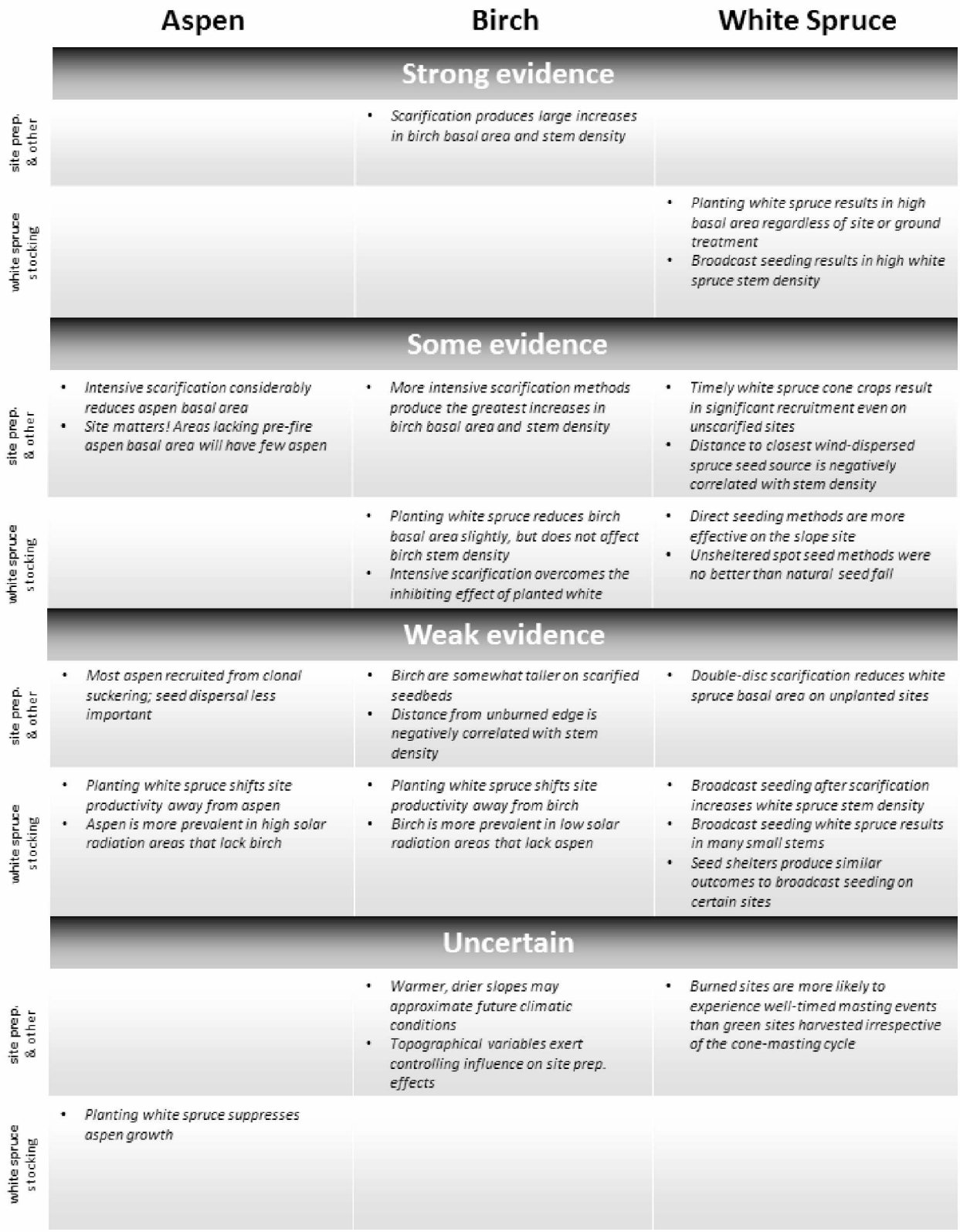

Figure 3.1: Management implications by species from study results.

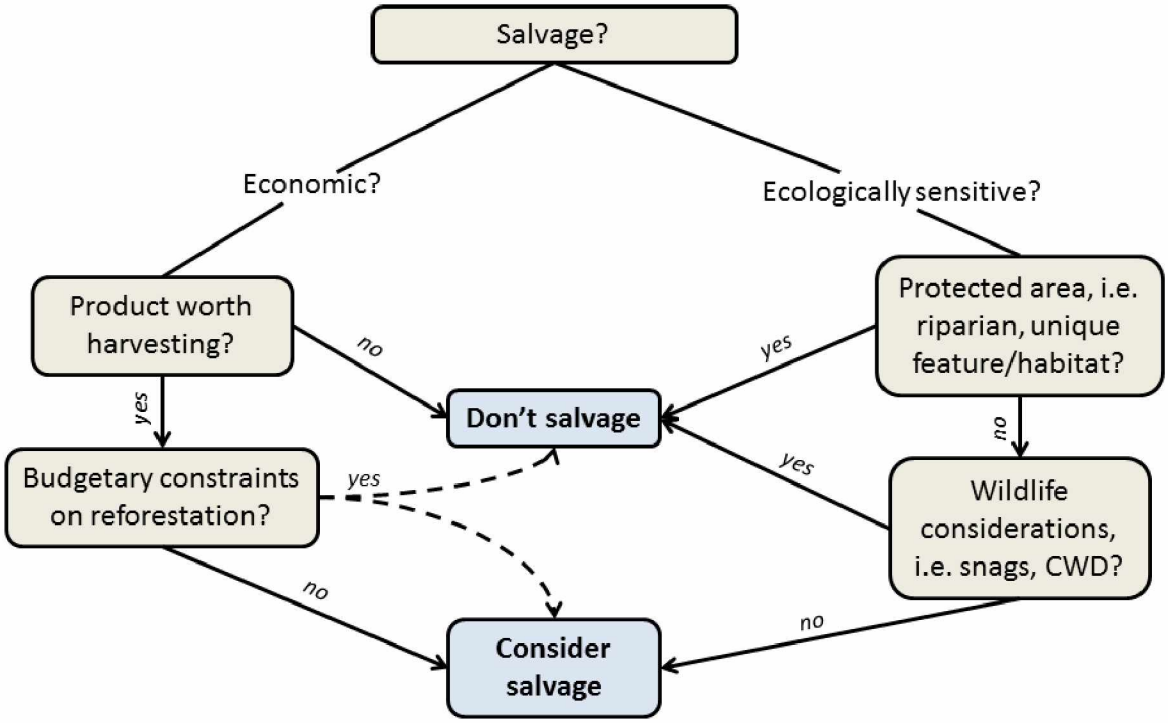

Figure 3.2: Decision tree whether to undertake salvage following a wildfire.

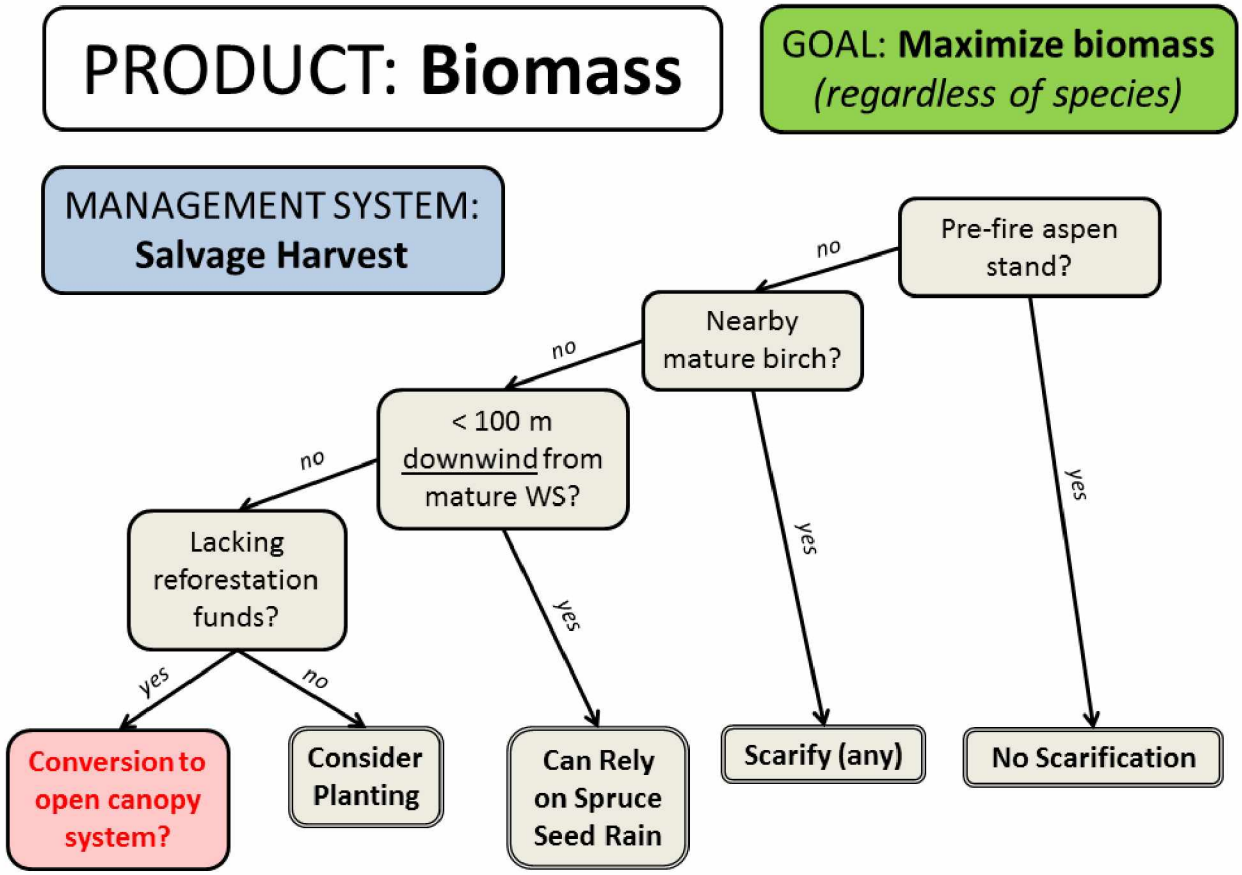

Figure 3.3: Decision tree for maximizing site biomass production.

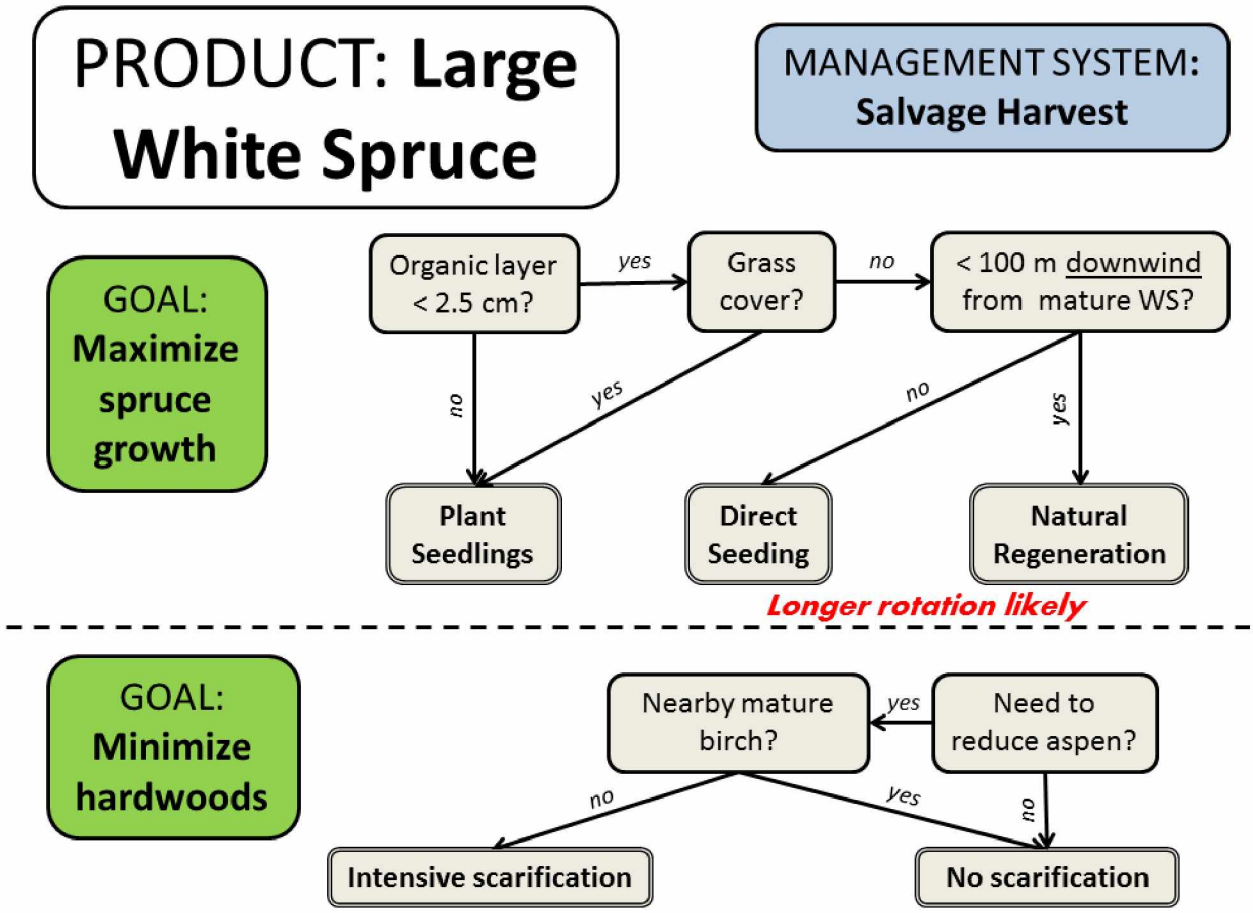

Figure 3.4: Decision tree for producing large-diameter white spruce.

Appendices

Appendix A

SAS command lines for full mixed model and ANOVA tables

Equation form for full mixed effects ANCOVA model.

$$Y_{ijk\ell} = \mu + \rho_\ell + \alpha_i + (\rho\alpha)_{i\ell} + \beta_j + (\alpha\beta)_{ij} + (\alpha\beta\rho)_{ij\ell} + \gamma_k + (\alpha\gamma)_{ik} + (\beta\gamma)_{jk} + (\alpha\beta\gamma)_{ijk} + \Theta_c * (x_{c,ijk\ell} - \overline{x}_{c,....}) + \varepsilon_{ijk\ell}$$

$Y_{ijk\ell}$ = dependent variable of interest on split- split plot ijkℓ

μ = grand mean of split-split plots

ρ_ℓ = *random* block effect (ℓ= 3)

α_i = fixed LF effect (i=2)

$\rho\alpha_{i\ell}$ = whole plot error

β_j = fixed GST effect (j=5)

$\alpha\beta_{ij}$ = fixed interaction, GST *LF effect

$\alpha\beta\rho_{ij\ell}$ = split plot error

γ_k = fixed plant effect (k=3)

$\alpha\gamma_{ik}$ = fixed interaction, LF*WSRT effect

$\beta\gamma_{jk}$ = fixed interaction, GST *WSRT effect

$\alpha\beta\gamma_{ijk}$ = fixed interaction, LF* GST *WSRT effect

Θ_c = regression coefficient for covariate *c*

$x_{c,ijk\ell}$ = covariate x_c measured on the split-split plot

$\overline{x}_{c,....}$ = mean of the covariate x_c for the entire experiment

$\varepsilon_{ijk\ell}$ = individual (split-split) plot error

Total Biomass

```
data rcftri3P;
set rcftri;
if (plant = "P") OR (plant = "C") OR (plant = "B");
run;

data rcftri3P_tot_massT;
set rcftri3P;
tot_massT=tot_mass**0.72;
run;

proc glm data=rcftri3P_tot_massT;
class  ground block plant site;
model tot_massT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
estimate "plant vs others" plant .5 .5 -1;
output out=glm3
predicted=fit residual=resid
rstudent=student;
run;
```

```
Dependent Variable: tot_massT
```

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	20310377.18	414497.49	2.17	0.0065
Error	40	7637825.20	190945.63		
Corrected Total	89	27948202.38			

R-Square	Coeff Var	Root MSE	tot_massT Mean
0.726715	13.34125	436.9733	3275.354

Source	DF	Type I SS	Mean Square	F Value	Pr > F
site	1	2022844.698	2022844.698	10.59	0.0023
block	2	950105.033	475052.517	2.49	0.0959
block*site	2	334699.326	167349.663	0.88	0.4241
ground	4	6181643.365	1545410.841	8.09	<.0001
ground*site	4	511410.686	127852.671	0.67	0.6169
ground*block*site	16	5958469.616	372404.351	1.95	0.0440
plant	2	153655.332	76827.666	0.40	0.6714
plant*site	2	341486.526	170743.263	0.89	0.4170
ground*plant	8	1572829.107	196603.638	1.03	0.4304
ground*plant*site	8	2283233.491	285404.186	1.49	0.1899

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	2022844.698	2022844.698	12.09	0.0737
block	2	950105.033	475052.517	2.84	0.2605

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	6181643.365	1545410.841	4.15	0.0171
ground*site	4	511410.686	127852.671	0.34	0.8447

Parameter	Estimate	Standard Error	t Value	Pr > |t|
scar vs unscar	593.860438	115.152565	5.16	<.0001
plant vs others	-75.117894	97.710191	-0.77	0.4465

Total Stem Density

```
data rcftri3P_tot_stemT;
set rcftri3P;
tot_stemT=tot_stem**0.45;
run;

proc glm data=rcftri3P_tot_stemT;
class  ground block plant site;
model tot_stemT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
estimate "broadcast vs others" plant -1 .5 .5;
output out=glm3
predicted=fit residual=resid
rstudent=student;
run;
```

Dependent Variable: tot_stemT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	11469.81169	234.07779	3.89	<.0001
Error	40	2406.50741	60.16269		
Corrected Total	89	13876.31910			

R-Square	Coeff Var	Root MSE	tot_stemT Mean
0.826575	13.21598	7.756461	58.69003

Source	DF	Type I SS	Mean Square	F Value	Pr > F
site	1	1180.591666	1180.591666	19.62	<.0001
block	2	34.825841	17.412921	0.29	0.7502

block*site	2	3567.561621	1783.780810	29.65	<.0001
ground	4	2045.745481	511.436370	8.50	<.0001
ground*site	4	519.458950	129.864738	2.16	0.0913
ground*block*site	16	3090.358987	193.147437	3.21	0.0014
plant	2	480.627501	240.313750	3.99	0.0262
plant*site	2	133.253729	66.626865	1.11	0.3403
ground*plant	8	166.321860	20.790233	0.35	0.9423
ground*plant*site	8	251.066054	31.383257	0.52	0.8330

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	1180.591666	1180.591666	0.66	0.5014
block	2	34.825841	17.412921	0.01	0.9903

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	2045.745481	511.436370	2.65	0.0719
ground*site	4	519.458950	129.864738	0.67	0.6207

Parameter	Estimate	Standard Error	t Value	Pr > |t|
scar vs unscar	10.5155230	2.04400691	5.14	<.0001
broadcast vs others	-4.6678582	1.73439738	-2.69	0.0103

Aspen Basal Area

```
data rcftri3P_A_baT;
set rcftri3P;
A_baT=log(A_ba+1);
run;

proc glm data=rcftri3P_A_baT;
class  ground block plant site;
model A_baT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
estimate "heavy scar vs unscar" ground .5 -1 .5 0 0;
estimate "heavy scar vs lt scar" ground -.5 0 -.5 .5 .5;
estimate "slope: heavy scar vs unscar" ground .5 -1 .5 0 0 site*ground 0 .5 0
-1 0 .5 0 0 0 0;
estimate "ridge: heavy scar vs unscar" ground .5 -1 .5 0 0 site*ground .5 0 -
1 0 .5 0 0 0 0 0;
estimate "slope: scar vs unscar" ground .25 -1 .25 .25 .25 site*ground 0 .25
0 -1 0 .25 0 .25 0 .25;
estimate "ridge: scar vs unscar" ground .25 -1 .25 .25 .25 site*ground .25 0
-1 0 .25 0 .25 0 .25 0;
estimate "slope: heavy scar vs lt scar" ground .5 0 .5 -.5 -.5 site*ground 0
.5 0 0 0 .5 0 -.5 0 -.5;
estimate "ridge: heavy scar vs lt scar" ground .5 0 .5 -.5 -.5 site*ground .5
0 0 0 .5 0 -.5 0 -.5 0;
estimate "slope: lt scar vs unscar" ground 0 -1 0 .5 .5 site*ground 0 0 0 -1
0 0 0 .5 0 .5;
estimate "ridge: lt scar vs unscar" ground 0 -1 0 .5 .5 site*ground 0 0 -1 0
0 0 .5 0 .5 0;
estimate "plant vs control" plant 0 1 -1;
estimate "ridge: plant vs control" plant 0 1 -1 site*plant 0 0 1 0 -1 0;
estimate "slope: plant vs control" plant 0 1 -1 site*plant 0 0 0 1 0 -1;
estimate "discing vs blade" ground -1 0 .5 0 .5;
estimate "discing vs control" ground 0 -1 .5 0 .5;
estimate "blade vs control" ground 1 -1 0 0 0;
estimate "lt scar vs unscar" ground 0 -1 0 .5 .5;
output out=glm3
predicted=fit residual=resid
```

```
rstudent=student;
run;
```

Dependent Variable: A_baT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	77.9475022	1.5907654	2.14	0.0073
Error	40	29.7356319	0.7433908		
Corrected Total	89	107.6831341			

R-Square	Coeff Var	Root MSE	A_baT Mean
0.723860	84.36892	0.862201	1.021942

Source	DF	Type I SS	Mean Square	F Value	Pr > F
site	1	7.55778062	7.55778062	10.17	0.0028
block	2	6.38678141	3.19339070	4.30	0.0204
block*site	2	1.57167866	0.78583933	1.06	0.3570
ground	4	5.36552892	1.34138223	1.80	0.1470
ground*site	4	16.17269532	4.04317383	5.44	0.0014
ground*block*site	16	26.30457076	1.64403567	2.21	0.0213
plant	2	1.19548100	0.59774050	0.80	0.4546
plant*site	2	0.96476630	0.48238315	0.65	0.5280
ground*plant	8	7.60867482	0.95108435	1.28	0.2815
ground*plant*site	8	4.81954440	0.60244305	0.81	0.5975

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	7.55778062	7.55778062	9.62	0.0901
block	2	6.38678141	3.19339070	4.06	0.1975

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	5.36552892	1.34138223	0.82	0.5335
ground*site	4	16.17269532	4.04317383	2.46	0.0875

Parameter	Estimate	Standard Error	t Value	Pr > |t|
scar vs unscar	-0.32944698	0.22720995	-1.45	0.1549
heavy scar vs unscar	-0.55399204	0.24889603	-2.23	0.0317
heavy scar vs lt scar	0.44909012	0.20322276	2.21	0.0329
slope: heavy scar vs unscar	-1.13608928	0.35199214	-3.23	0.0025
ridge: heavy scar vs unscar	0.02810520	0.35199214	0.08	0.9368

slope: scar vs unscar	-0.73173481	0.32132339	-2.28	0.0282
ridge: scar vs unscar	0.07284084	0.32132339	0.23	0.8218
slope: heavy scar vs lt scar	-0.80870895	0.28740038	-2.81	0.0076
ridge: heavy scar vs lt scar	-0.08947128	0.28740038	-0.31	0.7572
slope: lt scar vs unscar	-0.32738033	0.35199214	-0.93	0.3579
ridge: lt scar vs unscar	0.11757648	0.35199214	0.33	0.7401
plant vs control	0.25733359	0.22261938	1.16	0.2546
ridge: plant vs control	0.01324728	0.31483134	0.04	0.9666
slope: plant vs control	0.50141989	0.31483134	1.59	0.1191
discing vs blade	0.29899315	0.24889603	1.20	0.2367
discing vs control	-0.31570012	0.24889603	-1.27	0.2120
blade vs control	-0.61469327	0.28740038	-2.14	0.0386
lt scar vs unscar	-0.10490192	0.24889603	-0.42	0.6757

Aspen Stem Density

```
data rcftri3P_A_stemT;
set rcftri3P;
A_stemT=log(A_stem+1);
run;

proc glm data=rcftri3P_A_stemT;
class  ground block plant site;
model A_stemT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
estimate "heavy scar vs unscar" ground .5 -1 .5 0 0;
estimate "heavy scar vs lt scar" ground -.5 0 -.5 .5 .5;
estimate "slope: heavy scar vs unscar" ground .5 -1 .5 0 0 site*ground 0 .5 0
-1 0 .5 0 0 0 0;
estimate "ridge: heavy scar vs unscar" ground .5 -1 .5 0 0 site*ground .5 0 -
1 0 .5 0 0 0 0 0;
estimate "slope: scar vs unscar" ground .25 -1 .25 .25 .25 site*ground 0 .25
0 -1 0 .25 0 .25 0 .25;
estimate "ridge: scar vs unscar" ground .25 -1 .25 .25 .25 site*ground .25 0
-1 0 .25 0 .25 0 .25 0;
estimate "slope: heavy scar vs lt scar" ground .5 0 .5 -.5 -.5 site*ground 0
.5 0 0 0 .5 0 -.5 0 -.5;
estimate "ridge: heavy scar vs lt scar" ground .5 0 .5 -.5 -.5 site*ground .5
0 0 0 .5 0 -.5 0 -.5 0;
estimate "slope: lt scar vs unscar" ground 0 -1 0 .5 .5 site*ground 0 0 0 -1
0 0 0 .5 0 .5;
estimate "ridge: lt scar vs unscar" ground 0 -1 0 .5 .5 site*ground 0 0 -1 0
0 0 .5 0 .5 0;
estimate "plant vs control" plant 0 1 -1;
estimate "ridge: plant vs control" plant 0 1 -1 site*plant 0 0 1 0 -1 0;
estimate "slope: plant vs control" plant 0 1 -1 site*plant 0 0 0 1 0 -1;
estimate "discing vs blade" ground -1 0 .5 0 .5;
estimate "discing vs control" ground 0 -1 .5 0 .5;
estimate "blade vs control" ground 1 -1 0 0 0;
estimate "lt scar vs unscar" ground 0 -1 0 .5 .5;
output out=glm3
predicted=fit residual=resid
rstudent=student;
```

```
run;
```

Dependent Variable: A_stemT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	587.8595913	11.9971345	1.44	0.1178
Error	40	332.8677694	8.3216942		
Corrected Total	89	920.7273607			

R-Square	Coeff Var	Root MSE	A_stemT Mean
0.638473	77.91558	2.884735	3.702385

Source	DF	Type I SS	Mean Square	F Value	Pr > F
site	1	47.0403068	47.0403068	5.65	0.0223
block	2	50.9484265	25.4742132	3.06	0.0579
block*site	2	7.1282252	3.5641126	0.43	0.6546
ground	4	80.1417031	20.0354258	2.41	0.0653
ground*site	4	137.3092799	34.3273200	4.13	0.0068
ground*block*site	16	155.3788209	9.7111763	1.17	0.3338
plant	2	0.7610872	0.3805436	0.05	0.9554
plant*site	2	9.1942834	4.5971417	0.55	0.5799
ground*plant	8	50.6891411	6.3361426	0.76	0.6381
ground*plant*site	8	49.2683173	6.1585397	0.74	0.6559

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	47.04030676	47.04030676	13.20	0.0681
block	2	50.94842645	25.47421323	7.15	0.1227

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	80.1417031	20.0354258	2.06	0.1336
ground*site	4	137.3092799	34.3273200	3.53	0.0300

Parameter	Estimate	Standard Error	t Value	Pr > |t|
scar vs unscar	-1.55769797	0.76019434	-2.05	0.0471
heavy scar vs unscar	-2.22251479	0.83275118	-2.67	0.0109
heavy scar vs lt scar	1.32963364	0.67993849	1.96	0.0575
slope: heavy scar vs unscar	-3.83507926	1.17768801	-3.26	0.0023
ridge: heavy scar vs unscar	-0.60995032	1.17768801	-0.52	0.6074
slope: scar vs unscar	-2.80879957	1.07507714	-2.61	0.0126

ridge: scar vs unscar	-0.30659637	1.07507714	-0.29	0.7770
slope: heavy scar vs lt scar	-2.05255937	0.96157823	-2.13	0.0390
ridge: heavy scar vs lt scar	-0.60670790	0.96157823	-0.63	0.5317
slope: lt scar vs unscar	-1.78251988	1.17768801	-1.51	0.1380
ridge: lt scar vs unscar	-0.00324242	1.17768801	-0.00	0.9978
plant vs control	-0.17288079	0.74483529	-0.23	0.8176
ridge: plant vs control	-0.95568348	1.05335617	-0.91	0.3697
slope: plant vs control	0.60992190	1.05335617	0.58	0.5658
discing vs blade	0.00000812	0.83275118	0.00	1.0000
discing vs control	-1.96320590	0.83275118	-2.36	0.0234
blade vs control	-1.96321402	0.96157823	-2.04	0.0478
lt scar vs unscar	-0.89288115	0.83275118	-1.07	0.2901

Birch Basal Area

```
data rcftri3P_B_baT;
set rcftri3P;
B_baT=B_ba**0.57;
run;

proc glm data=rcftri3P;
class  ground block plant site;
model B_stemT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "planting vs others" plant .5 .5 -1;
estimate "planting: scar vs unscar" ground 0.25 -1 0.25 0.25 0.25
ground*plant 0 0 0.25 0 0 -1 0 0 0.25 0 0 0.25 0 0 0.25;
estimate "planting: heavy scar vs  lt scar" ground -0.5 0 -0.5 0.5 0.5
ground*plant 0 0 -0.5 0 0 0 0 0 -0.5 0 0 0.5 0 0 0.5;
estimate "heavy scar vs lt" ground .5 0 .5 -.5 -.5;
estimate "heavy scar vs unscar" ground .5 -1 .5 0 0;
estimate "lt scar vs unscar" ground 0 -1 0 .5 .5;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
estimate "heavy scar: plant vs control" plant 0 1 -1 ground*plant 0 .5 -.5 0
0 0 0 .5 -.5 0 0 0 0 0 0;
output out=glm3
predicted=fit residual=resid
rstudent=student;
run;
```

Dependent Variable: B_baT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	246.6666111	5.0340125	9.71	<.0001
Error	40	20.7301657	0.5182541		
Corrected Total	89	267.3967768			

R-Square	Coeff Var	Root MSE	B_baT Mean

0.922474	18.64477	0.719899	3.861129	

Source	DF	Type I SS	Mean Square	F Value	Pr > F
site	1	90.17326520	90.17326520	173.99	<.0001
block	2	9.75969703	4.87984851	9.42	0.0004
block*site	2	2.51101758	1.25550879	2.42	0.1016
ground	4	36.29148646	9.07287162	17.51	<.0001
ground*site	4	17.30401818	4.32600455	8.35	<.0001
ground*block*site	16	70.87497683	4.42968605	8.55	<.0001
plant	2	3.47688749	1.73844374	3.35	0.0450
plant*site	2	0.42039476	0.21019738	0.41	0.6693
ground*plant	8	10.31545357	1.28943170	2.49	0.0271
ground*plant*site	8	5.53941399	0.69242675	1.34	0.2543

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	90.17326520	90.17326520	71.82	0.0136
block	2	9.75969703	4.87984851	3.89	0.2046

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	36.29148646	9.07287162	2.05	0.1358
ground*site	4	17.30401818	4.32600455	0.98	0.4476

Parameter	Estimate	Standard Error	t Value	Pr > |t|
planting vs others	0.45883471	0.30186444	1.52	0.1364
planting: scar vs unscar	3.73616944	0.61617820	6.06	<.0001
planting: heavy scar vs lt scar	-2.37475759	0.55112654	-4.31	0.0001
heavy scar vs lt	1.89047879	0.31819306	5.94	<.0001
heavy scar vs unscar	4.57991221	0.38970531	11.75	<.0001
lt scar vs unscar	2.68943342	0.38970531	6.90	<.0001
scar vs unscar	3.63467282	0.35575065	10.22	<.0001
heavy scar: plant vs control	0.07194291	0.55112654	0.13	0.8968

Birch Stem Density

```
proc glm data=rcftri3P_B_stemT;
class  ground block plant site;
model B_stemT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "planting: scar vs unscar" ground 0.25 -1 0.25 0.25 0.25
ground*plant 0 0 0.25 0 0 -1 0 0 0.25 0 0 0.25 0 0 0.25;
estimate "planting: heavy scar vs  lt scar" ground -0.5 0 -0.5 0.5 0.5
ground*plant 0 0 -0.5 0 0 0 0 0 -0.5 0 0 0.5 0 0 0.5;
estimate "heavy scar: plant vs control" plant 0 1 -1 ground*plant 0 .5 -.5 0
0 0 .5 -.5 0 0 0 0 0 0;
estimate "heavy scar vs lt" ground .5 0 .5 -.5 -.5;
estimate "heavy scar vs unscar" ground .5 -1 .5 0 0;
estimate "lt scar vs unscar" ground 0 -1 0 .5 .5;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
output out=glm3
predicted=fit residual=resid
rstudent=student;
run;
```

Dependent Variable: B_stemT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	1814.438449	37.029356	20.32	<.0001
Error	40	72.897711	1.822443		
Corrected Total	89	1887.336160			

R-Square	Coeff Var	Root MSE	B_stemT Mean
0.961375	9.187050	1.349979	14.69437

Source	DF	Type I SS	Mean Square	F Value	Pr > F

site	1	658.4537146	658.4537146	361.30	<.0001
block	2	3.1193628	1.5596814	0.86	0.4326
block*site	2	239.0172290	119.5086145	65.58	<.0001
ground	4	268.8251283	67.2062821	36.88	<.0001
ground*site	4	97.8822250	24.4705563	13.43	<.0001
ground*block*site	16	500.0629653	31.2539353	17.15	<.0001
plant	2	5.2691873	2.6345937	1.45	0.2476
plant*site	2	5.5339719	2.7669859	1.52	0.2314
ground*plant	8	24.7000111	3.0875014	1.69	0.1299
ground*plant*site	8	11.5746535	1.4468317	0.79	0.6111

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	658.4537146	658.4537146	5.51	0.1435
block	2	3.1193628	1.5596814	0.01	0.9871

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	268.8251283	67.2062821	2.15	0.1215
ground*site	4	97.8822250	24.4705563	0.78	0.5526

Parameter	Estimate	Standard Error	t Value	Pr > |t|
planting: scar vs unscar	3.73616944	0.61617820	6.06	<.0001
planting: heavy scar vs lt scar	-2.37475759	0.55112654	-4.31	0.0001
heavy scar: plant vs control	0.07194291	0.55112654	0.13	0.8968
heavy scar vs lt	1.89047879	0.31819306	5.94	<.0001
heavy scar vs unscar	4.57991221	0.38970531	11.75	<.0001
lt scar vs unscar	2.68943342	0.38970531	6.90	<.0001
scar vs unscar	3.63467282	0.35575065	10.22	<.0001

White Spruce Basal Area

```
data rcftri3P_WS_baT;
set rcftri3P;
WS_baT=WS_ba**0.23;
run;

proc glm data=rcftri3P_WS_baT;
class  ground block plant site;
model WS_baT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
estimate "planted: scar vs unscar" ground .25 -1 .25 .25 .25 ground*plant 0 0
.25 0 0 -1 0 0 .25 0 0 .25 0 0 .25;
estimate "broadcast: scar vs unscar" ground .25 -1 .25 .25 .25 ground*plant
.25 0 0 -1 0 0 .25 0 0 .25 0 0 .25 0 0;
estimate "plant: heavy scar vs unscar" ground .5 -1 .5 0 0 ground*plant 0 0
.5 0 0 -1 0 0 .5 0 0 0 0 0 0;
estimate "natural seedfall: scar vs unscar" ground .25 -1 .25 .25 .25
ground*plant 0 .25 0 0 -1 0 0 .25 0 0 .25 0 0 .25 0;
output out=glm3
predicted=fit residual=resid
rstudent=student;
run;
```

Dependent Variable: WS_baT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	15.44652375	0.31523518	6.44	<.0001
Error	40	1.95781776	0.04894544		
Corrected Total	89	17.40434152			

R-Square	Coeff Var	Root MSE	WS_baT Mean

0.887510 19.08738 0.221236 1.159070

Source	DF	Type I SS	Mean Square	F Value	Pr > F
site	1	4.13343669	4.13343669	84.45	<.0001
block	2	0.33616111	0.16808056	3.43	0.0420
block*site	2	1.84679059	0.92339530	18.87	<.0001
ground	4	0.11539063	0.02884766	0.59	0.6722
ground*site	4	0.36366361	0.09091590	1.86	0.1369
ground*block*site	16	0.89338358	0.05583647	1.14	0.3539
plant	2	6.81381820	3.40690910	69.61	<.0001
plant*site	2	0.37722746	0.18861373	3.85	0.0295
ground*plant	8	0.15320788	0.01915099	0.39	0.9188
ground*plant*site	8	0.41344401	0.05168050	1.06	0.4125

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	4.13343669	4.13343669	4.48	0.1686
block	2	0.33616111	0.16808056	0.18	0.8460

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	0.11539063	0.02884766	0.52	0.7247
ground*site	4	0.36366361	0.09091590	1.63	0.2157

Parameter	Estimate	Standard Error	t Value	Pr > |t|
scar vs unscar	-0.06290401	0.05830085	-1.08	0.2871
planted: scar vs unscar	-0.09210386	0.10098004	-0.91	0.3672
broadcast: scar vs unscar	-0.05234485	0.10098004	-0.52	0.6071
plant: heavy scar vs unscar	-0.10084946	0.11061809	-0.91	0.3674
natural seedfall: scar vs unscar	-0.04426333	0.10098004	-0.44	0.6635

White Spruce Stem Density

```
data rcftri3P_WS_stemT;
set rcftri3P;
WS_stemT=WS_stem**0.27;
run;

proc glm data=rcftri3P_WS_stemT;
class  ground block plant site;
model WS_stemT = site
                        block site*block
                        ground site*ground site*block*ground
                        plant site*plant ground*plant site*ground*plant;
random block site*block site*block*ground;
test h=site e=site*block;
test h=block e=site*block;
test h=ground e=site*block*ground;
test h=site*ground e=site*block*ground;
lsmeans ground site*ground / stderr pdiff cl adjust=tukey
e=ground*block*site;
lsmeans site block / pdiff cl adjust=tukey e=block*site;
lsmeans plant site*plant ground*plant site*ground*plant/ stderr pdiff cl
adjust=tukey;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans ground*plant/ pdiff=CONTROLU("C" "C") cl adjust=dunnett;
lsmeans ground site*ground / pdiff=CONTROLU("C") cl adjust=dunnett
e=ground*block*site;
lsmeans plant/ pdiff=CONTROLU("C") cl adjust=dunnett;
estimate "broadcast ridge: scar vs unscar" ground 0.25 -1 0.25 0.25 0.25
      site*ground .25 0 -1 0 .25 0 .25 0 .25 0
      ground*plant .25 0 0 -1 0 0 .25 0 0 .25 0 0 .25 0 0
      site*ground*plant .25 0 0 0 0 0 -1 0 0 0 0 0 .25 0 0 0 0 0 .25 0 0 0 0
0 .25 0 0 0 0 0;
estimate "broadcast slope: scar vs unscar" ground 0.25 -1 0.25 0.25 0.25
      site*ground 0 .25 0 -1 0 .25 0 .25 0 .25
      ground*plant .25 0 0 -1 0 0 .25 0 0 .25 0 0 .25 0 0
      site*ground*plant 0 .25 0 0 0 0 0 -1 0 0 0 0 0 .25 0 0 0 0 0 .25 0 0 0
0 0 .25 0 0 0 0;
estimate "planted: scar vs unscar" ground .25 -1 .25 .25 .25 ground*plant 0 0
.25 0 0 -1 0 0 .25 0 0 .25 0 0 .25;
estimate "ridge: control vs broadcast" plant 1 -1 0 site*plant 1 0 -1 0 0 0;
estimate "natural seedfall: scar vs unscar" ground .25 -1 .25 .25 .25
ground*plant 0 .25 0 0 -1 0 0 .25 0 0 .25 0 0 .25 0;
estimate "natural seedfall ridge: scar vs unscar" ground 0.25 -1 0.25 0.25
0.25
      site*ground .25 0 -1 0 .25 0 .25 0 .25 0
      ground*plant 0 .25 0 0 -1 0 0 .25 0 0 .25 0 0 .25 0
      site*ground*plant 0 0 .25 0 0 0 0 0 -1 0 0 0 0 0 .25 0 0 0 0 0 .25 0 0
0 0 0 .25 0 0 0;
estimate "natural seedfall slope: scar vs unscar" ground 0.25 -1 0.25 0.25
0.25
      site*ground 0 .25 0 -1 0 .25 0 .25 0 .25
      ground*plant 0 .25 0 0 -1 0 0 .25 0 0 .25 0 0 .25 0
      site*ground*plant 0 0 0 .25 0 0 0 0 0 -1 0 0 0 0 0 .25 0 0 0 0 0 .25 0
0 0 0 0 .25 0 0;
output out=glm3
```

```
predicted=fit residual=resid
rstudent=student;
run;
```

Dependent Variable: WS_stemT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	339.4515862	6.9275834	3.70	<.0001
Error	40	74.8175142	1.8704379		
Corrected Total	89	414.2691003			

R-Square	Coeff Var	Root MSE	WS_stemT Mean
0.819399	20.51713	1.367640	6.665844

Source	DF	Type I SS	Mean Square	F Value	Pr > F
site	1	120.9697296	120.9697296	64.67	<.0001
block	2	23.2090970	11.6045485	6.20	0.0045
block*site	2	23.9562623	11.9781312	6.40	0.0039
ground	4	1.2829996	0.3207499	0.17	0.9517
ground*site	4	11.1702233	2.7925558	1.49	0.2225
ground*block*site	16	31.5191602	1.9699475	1.05	0.4273
plant	2	80.8273329	40.4136665	21.61	<.0001
plant*site	2	23.0535887	11.5267944	6.16	0.0046
ground*plant	8	6.5240502	0.8155063	0.44	0.8922
ground*plant*site	8	16.9391423	2.1173928	1.13	0.3635

Tests of Hypotheses Using the Type III MS for block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
site	1	120.9697296	120.9697296	10.10	0.0864
block	2	23.2090970	11.6045485	0.97	0.5079

Tests of Hypotheses Using the Type III MS for ground*block*site as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	1.28299961	0.32074990	0.16	0.9541
ground*site	4	11.17022326	2.79255581	1.42	0.2732

Parameter	Estimate	Standard Error	t Value	Pr > |t|
broadcast ridge: scar vs unscar	-1.06456953	0.88280751	-1.21	0.2349
broadcast slope: scar vs unscar	2.04613909	0.88280751	2.32	0.0257
planted: scar vs unscar	-0.24373873	0.62423918	-0.39	0.6983
ridge: control vs broadcast	2.16183409	0.49939134	4.33	<.0001

natural seedfall: scar vs unscar	0.21592555	0.62423918	0.35	0.7312
natural seedfall ridge: scar vs unscar	0.41753093	0.88280751	0.47	0.6388
natural seedfall slope: scar vs unscar	0.01432017	0.88280751	0.02	0.9871

Appendix B

SAS command lines for reduced mixed model (site excluded) and ANOVA tables

White Spruce Basal Area

```
data rcftri3wholeblocks;
set rcftri;
if (sblock = "R1") OR (sblock = "R3") OR (sblock = "S1");
run;

data rcftri3wholeblocks_WS_baT;
set rcftri3wholeblocks;
WS_baT=WS_ba**0.21;
run;

proc glm data=rcftri3wholeblocks_WS_baT;
class sblock ground plant;
model WS_baT=sblock ground sblock*ground plant ground*plant sblock*plant;
random sblock sblock*ground;
test h=ground e=sblock*ground;
lsmeans sblock*ground ground / stderr pdiff cl adjust=tukey e=sblock*ground;
lsmeans plant ground*plant sblock*plant / stderr pdiff cl adjust=tukey;
estimate "shelter by site" sblock .5 .5 -1 sblock*plant 0 0 0 0 .5 0 0 0 0 0
.5 0 0 0 0 0 -1 0;
estimate "broadcast by site" sblock .5 .5 -1 sblock*plant .5 0 0 0 0 0 .5 0 0
0 0 0 -1 0 0 0 0 0;
output out=glm1
predicted=fit residual=resid
rstudent=studentized residual;
run;
```

Dependent Variable: WS_baT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	10.35206402	0.21126661	3.84	<.0001
Error	40	2.19938326	0.05498458		
Corrected Total	89	12.55144728			

R-Square	Coeff Var	Root MSE	WS_baT Mean
0.824771	23.81360	0.234488	0.984681

Source	DF	Type I SS	Mean Square	F Value	Pr > F
sblock	2	4.10162685	2.05081343	37.30	<.0001
ground	4	0.17605940	0.04401485	0.80	0.5321
sblock*ground	8	0.68158713	0.08519839	1.55	0.1712
plant	5	4.17717239	0.83543448	15.19	<.0001
ground*plant	20	0.70717358	0.03535868	0.64	0.8545
sblock*plant	10	0.50844467	0.05084447	0.92	0.5210

Tests of Hypotheses Using the Type III MS for sblock*ground as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	0.17605940	0.04401485	0.52	0.7264

Parameter	Estimate	Standard Error	t Value	Pr > |t|
shelter by site	-0.45837166	0.12843432	-3.57	0.0009
broadcast by site	-0.29023025	0.12843432	-2.26	0.0294

White Spruce Stem Density

```
data rcftri3wholeblocks_WS_stemT;
set rcftri3wholeblocks;
WS_stemT=WS_stem**0.27;
run;

proc glm data=rcftri3wholeblocks_WS_stemT;
class sblock ground plant;
model WS_stemT=sblock ground sblock*ground plant ground*plant sblock*plant;
random sblock sblock*ground;
test h=ground e=sblock*ground;
lsmeans sblock*ground ground / stderr pdiff cl adjust=tukey e=sblock*ground;
lsmeans plant ground*plant sblock*plant / stderr pdiff cl adjust=tukey;
estimate "scar vs unscar" ground .25 -1 .25 .25 .25;
output out=glm1
predicted=fit residual=resid
rstudent=studentized residual;
run;
```

Dependent Variable: WS_stemT

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	49	359.7585101	7.3420104	3.99	<.0001
Error	40	73.6305335	1.8407633		
Corrected Total	89	433.3890436			

R-Square	Coeff Var	Root MSE	WS_stemT Mean
0.830105	21.67303	1.356747	6.260073

Source	DF	Type I SS	Mean Square	F Value	Pr > F
sblock	2	181.8911216	90.9455608	49.41	<.0001
ground	4	4.6223138	1.1555785	0.63	0.6455
sblock*ground	8	42.0649398	5.2581175	2.86	0.0131
plant	5	66.9813794	13.3962759	7.28	<.0001
ground*plant	20	38.3547172	1.9177359	1.04	0.4410
sblock*plant	10	25.8440383	2.5844038	1.40	0.2138

Tests of Hypotheses Using the Type III MS for sblock*ground as an Error Term

Source	DF	Type III SS	Mean Square	F Value	Pr > F
ground	4	4.62231384	1.15557846	0.22	0.9200

Parameter	Estimate	Standard Error	t Value	Pr > |t|
scar vs unscar	-0.48318160	0.35753432	-1.35	0.1842

Appendix C

Supplemental Figures and Tables

Tree Height Data Collection

Table C.1: Count of recorded heights across species and experimental treatments.

		Aspen	Birch	White Spruce	Total
WSRT	**Planted Seedling**	57	90	97	244
	Broadcast	15	71	37	123
	Control/Spot Seed	41	179	96	316
Ground Treatment	**Scarified**	49	164	96	309
	Non-scarified	64	176	134	374
Site	**Ridge**	31	121	68	220
	Slope-Upper	57	122	98	277
	Slope-Lower	25	97	64	186
	All sites	113	340	230	683

Species Stem Density and Biomass by Site

Table C.2: Stem density and aboveground biomass averaged across non-scarified, natural seedfall control S/SPs (n=6) by landform whole plot. Totals are weighted by sample area, for the 135 sampled S/SPs.

	STEM DENSITY (stems ha^{-1})			ABOVEGROUND BIOMASS (kg ha^{-1})		
	Ridge	**Slope**	**Total**	**Ridge**	**Slope**	**Total**
Aspen	121	825	514	7,379	25,042	17,250
Alder	534	1,227	922	704	1,694	1,257
Birch	5,344	3,567	4,351	62,326	16,657	36,805
Willow	73	326	214	48	478	288
White Spruce	340	2,071	1,307	1,130	7,745	4,827
Total	6,412	8,015	7,308	71,588	51,616	60,427

Pie Charts of Species Composition by Landform Whole Plot

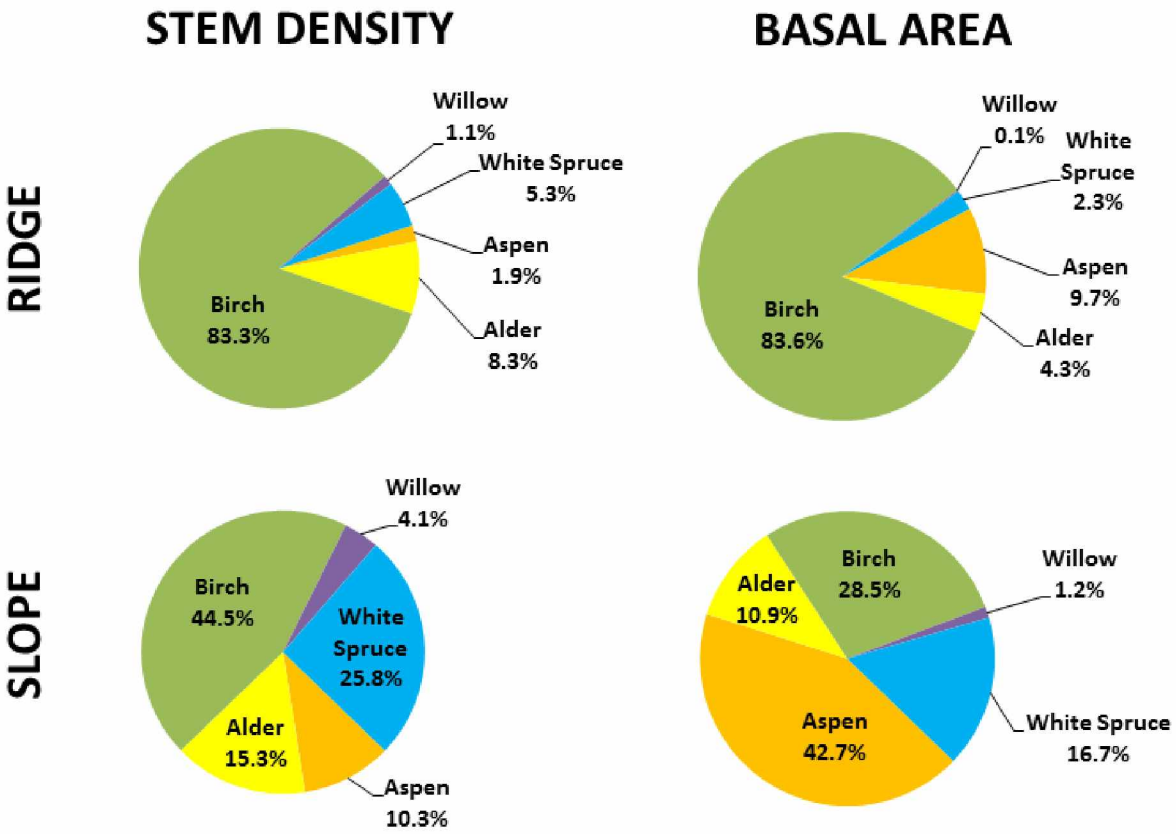

Figure C.1: Species composition by stem density and basal area on slope and ridge LFs for non-scarified, natural seedfall control S/SPs (n=6).

Census Results

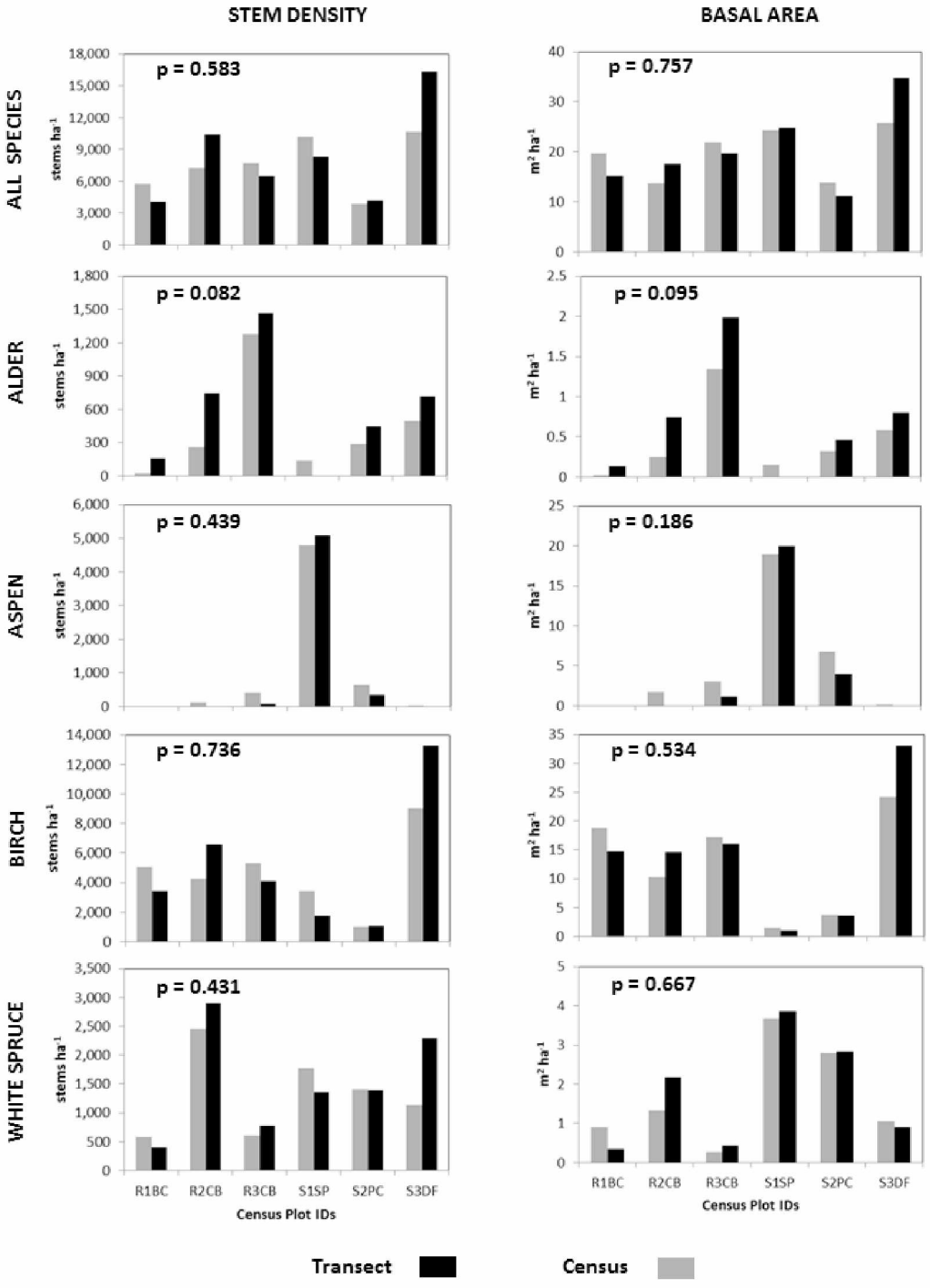

Figure C.2: Comparison of census and transect results for six S/SPs. P-values for paired t-tests are displayed on each graph, where the null hypothesis is that the census and transect means are the same. S/SPs include R1BC (Ridge, Block 1, Blade GST, natural seedfall Control WSRT), R2CB (Ridge, Block 2, non-scarified Control GST, Broadcast WSRT), R3CB (Ridge, Block 2, non-scarified Control GST, Broadcast WSRT), S1SP (Slope, Block 1, Single-disc GST, Planted seedling WSRT), S2PC (Slope, Block 2, Patch GST, natural seedfall Control WSRT), and S3DF (Slope, Block 3, Double-disc GST, Fall spot seed WSRT).

LTER Weather Data

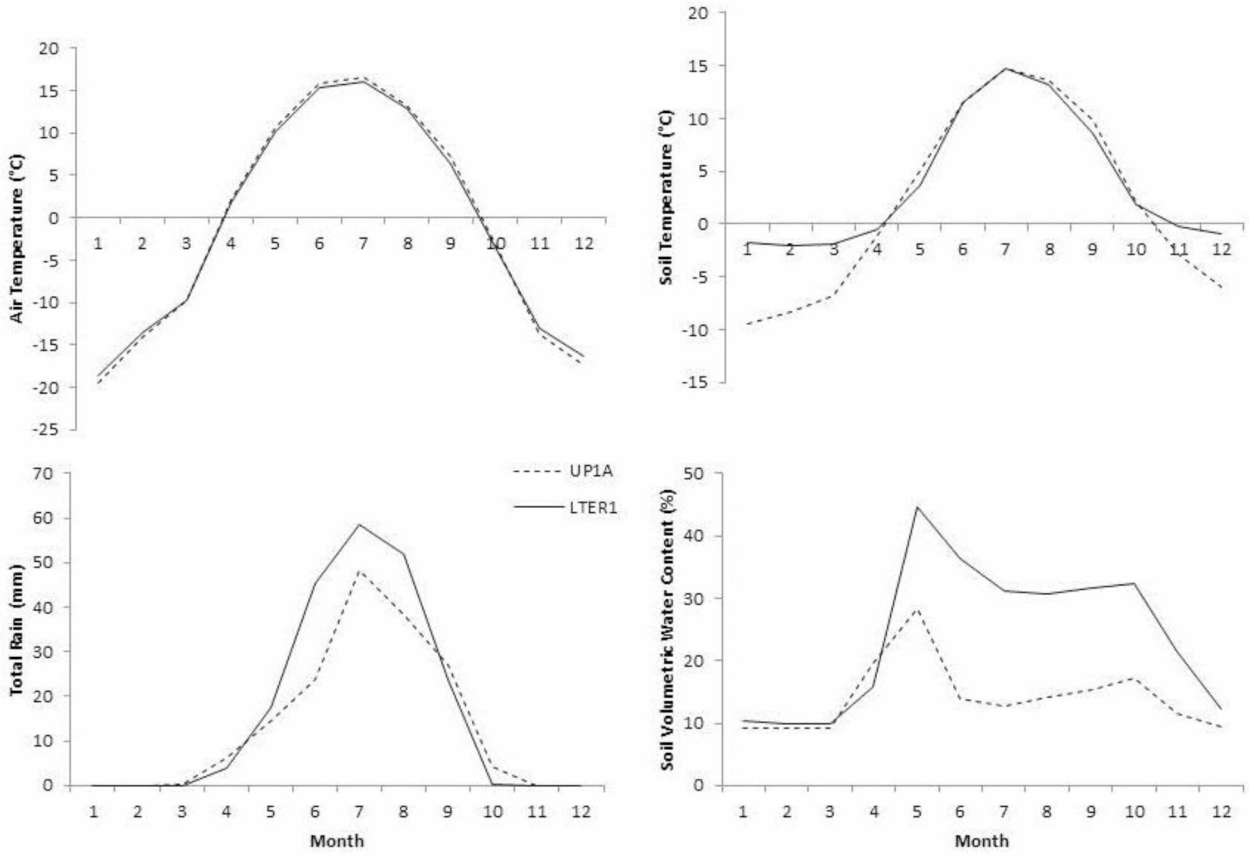

Figure C.3: Climate graphs with mean monthly observations for two weather stations. These derive from the nearby mid-slope UP1A second-order weather station and the ridge top LTER1 first order weather station near to RCFTRI.

Covariate Analysis Tables by Species

Table C.3: Covariate analysis for total stem density and total biomass. A significant p-value (< 0.05; highlighted) for the linear regression of covariate and response was subsequently tested for inclusion in the full ANCOVA model to reduce experimental variance. Pearson correlation coefficient and model R^2 from the simple linear regression are displayed.

	TOTAL STEM DENSITY 90 S/SPs, response=(stems ha^{-1})$^{0.45}$				**TOTAL MASS** 90 S/SPs, response=(kg ha^{-1})$^{0.72}$			
	Simple Linear Regression			**ANCOVA**	**Simple Linear Regression**			**ANCOVA**
covariate	p-value	correl.	R^2	p-value	p-value	correl.	R^2	p-value
Elevation	0.0011	0.34	0.11	0.7875	0.0089	0.27	0.08	0.1561
Slope	0.6722	-0.05	0.00	N/A	0.0273	-0.23	0.05	0.6749
Solar	0.5152	-0.07	0.00	N/A	0.0117	-0.26	0.07	0.3918
Fire Distance	0.0001	-0.40	0.16	0.5531	0.0006	-0.35	0.12	0.9450
ln(Fire Distance)	0.0002	-0.38	0.14	0.6678	0.0009	-0.34	0.12	0.9595
Closest Seed	0.1183	-0.17	0.03	N/A	0.0364	-0.22	0.05	0.5541
Mean Seed	0.9761	0.00	0.00	N/A	0.2217	0.13	0.02	N/A

Table C.4: Covariate analysis for white spruce stem density and basal area. A significant p-value (< 0.05; highlighted) for the linear regression of covariate and response was subsequently tested for inclusion in the full ANCOVA model to reduce experimental variance. Pearson correlation coefficient and model R^2 from the simple linear regression are displayed. White spruce is explicitly affected by WSRT, so we also evaluated the subset of S/SPs (n=30) on which only natural seedfall was allowed.

	WHITE SPRUCE STEM DENSITY 90 S/SPs, response=(stems ha^{-1})$^{0.27}$				**WHITE SPRUCE BASAL AREA** 90 S/SPs, response=(m^2 ha^{-1})$^{0.23}$			
	Simple Linear Regression			**ANCOVA**	**Simple Linear Regression**			**ANCOVA**
covariate	p-value	correl.	R^2	p-value	p-value	correl.	R^2	p-value
Elevation	0.0001	-0.44	0.19	0.6683	0.0001	-0.43	0.18	0.2970
Slope	0.0008	0.35	0.12	0.1848	0.0028	0.31	0.10	0.2449
Solar	0.8604	0.02	0.00	N/A	0.3224	0.11	0.01	N/A
Fire Distance	0.7452	0.03	0.00	N/A	0.1133	0.17	0.03	N/A
ln(Fire Dist.)	0.9544	0.01	0.00	N/A	0.1907	0.14	0.02	N/A
Closest Seed	0.0133	-0.26	0.07	0.9744	0.2697	-0.12	0.01	0.7199
Mean Seed	0.0001	-0.45	0.21	0.6495	0.0017	-0.33	0.11	0.7473
ln(Spruce Dist.)	0.0001	-0.42	0.17	0.2052				

	WHITE SPRUCE STEM DENSITY 30 S/SPs, response=(stems ha^{-1})$^{0.20}$			**WHITE SPRUCE BASAL AREA** 30 S/SPs, response=(m^2 ha^{-1})$^{0.20}$		
	Simple Linear Regression			**Simple Linear Regression**		
covariate	p-value	correl.	R^2	p-value	correl.	R^2
Elevation	0.0081	-0.47	0.23	0.0015	-0.55	0.31
Slope	0.0092	0.47	0.22	0.0034	0.52	0.27
Solar	0.7642	-0.06	0.00	0.9110	0.02	0.00
Fire Distance	0.8050	-0.05	0.00	0.7445	0.06	0.00
ln(Fire Dist.)	0.6661	-0.08	0.01	0.8463	0.04	0.00
Closest Seed	0.0089	-0.47	0.22	0.0441	-0.37	0.14
Mean Seed	0.0200	-0.42	0.18	0.0245	-0.41	0.17
ln(Spruce Dist.)	0.0056	-0.49	0.24			

Table C.5: Covariate analysis for birch stem density and basal area. A significant p-value (< 0.05; highlighted) for the linear regression of covariate and response was subsequently tested for inclusion in the full ANCOVA model to reduce experimental variance. Pearson correlation coefficient and model R^2 from the simple linear regression are displayed. Note that closest seed and mean seed have opposite correlations.

	BIRCH SPRUCE STEM DENSITY 90 S/SPs, response=(stems ha^{-1})$^{0.32}$				**BIRCH SPRUCE BASAL AREA** 90 S/SPs, response=(m^2 ha^{-1})$^{0.57}$			
	Simple Linear Regression			**ANCOVA**	**Simple Linear Regression**			**ANCOVA**
covariate	p-value	correl.	R^2	p-value	p-value	correl.	R^2	p-value
Elevation	0.0001	0.63	0.39	0.6570	0.0001	0.54	0.29	0.1025
Slope	0.0037	-0.30	0.09	0.7352	0.0002	-0.38	0.15	0.6781
Solar	0.1379	-0.16	0.02	N/A	0.0277	-0.23	0.05	0.9453
Fire Distance	0.0001	-0.53	0.29	0.7023	0.0001	-0.46	0.22	0.9855
ln(Fire Distance)	0.0001	-0.48	0.23	0.1462	0.0001	-0.41	0.17	0.9045
Closest Seed	0.0254	-0.24	0.06	0.8584	0.0227	-0.24	0.06	0.9941
Mean Seed	0.0235	0.24	0.06	0.6068	0.0002	0.38	0.14	0.0960

Table C.6: Covariate analysis for aspen stem density and basal area. A significant p-value (< 0.05; highlighted) for the linear regression of covariate and response was subsequently tested for inclusion in the full ANCOVA model to reduce experimental variance. A number of covariates, though not significant at the p<0.05 level, were nearly so and therefore tested for inclusion in the full model. Pearson correlation coefficient and model R^2 from the simple linear regression are displayed.

	ASPEN STEM DENSITY 90 S/SPs, response=ln (stems ha^{-1} + 1)				**ASPEN SPRUCE BASAL AREA** 90 S/SPs, response=ln (m^2 ha^{-1} + 1)			
	Simple Linear Regression			**ANCOVA**	**Simple Linear Regression**			**ANCOVA**
covariate	p-value	correl.	R^2	p-value	p-value	correl.	R^2	p-value
Elevation	0.1315	-0.16	0.03	N/A	0.0551	-0.20	0.04	0.0761
Slope	0.0003	0.37	0.14	0.1164	0.0021	0.32	0.10	0.1732
Solar	0.0048	0.29	0.09	0.9034	0.0342	0.22	0.05	0.8857
Fire Distance	0.0538	0.20	0.04	0.0663	0.0465	0.21	0.04	0.2429
ln(Fire Distance)	0.0518	0.21	0.04	0.4306	0.0621	0.20	0.04	0.6845
Closest Seed	0.4677	0.08	0.01	N/A	0.4103	0.09	0.01	N/A
Mean Seed	0.1258	-0.16	0.03	N/A	0.0606	-0.20	0.04	0.9718

Aspen stem density with additional split-split plots (S/SPs)

The effects of scarification on aspen stem density were obscured by outliers, particularly on the low density but highly variable ridge site. For example, one influential lightly scarified S/SP in Ridge Block 2 had an estimate of 5,660 stems ha^{-1}. Aspen on the ridge whole plot responded quite differently to scarification than on the slope, averaging 503 ± 313 stems ha^{-1} on lightly scarified sites, 209 ± 75 stems ha^{-1} on intensively scarified S/SPs, and 90 ± 27 stems ha^{-1} on non-scarified S/SPs. Since WSRT was not a significant factor determining aspen density, we expanded our ridge dataset to 75 plots and our slope dataset to 60 S/SPs. These additional S/SPs included Ridge Block 1, Ridge Block 3, and Slope Block 1 for all six WSRTs. This expanded sample is balanced across the five GSTs.

For the slope none of the findings change significantly with the expanded sample size, but now the ridge approximates similar results. Furthermore, contrasts between intensive scarification and non-scarified S/SPs is marginally significant on the ridge (two linear contrasts: with n=45, p=0.6074 compared to n=75, p=0.0509), and similar results for intensive scarification compared to non-scarified S/SPs (with n=45, p=0.5317 compared to n=75, p=0.0606).

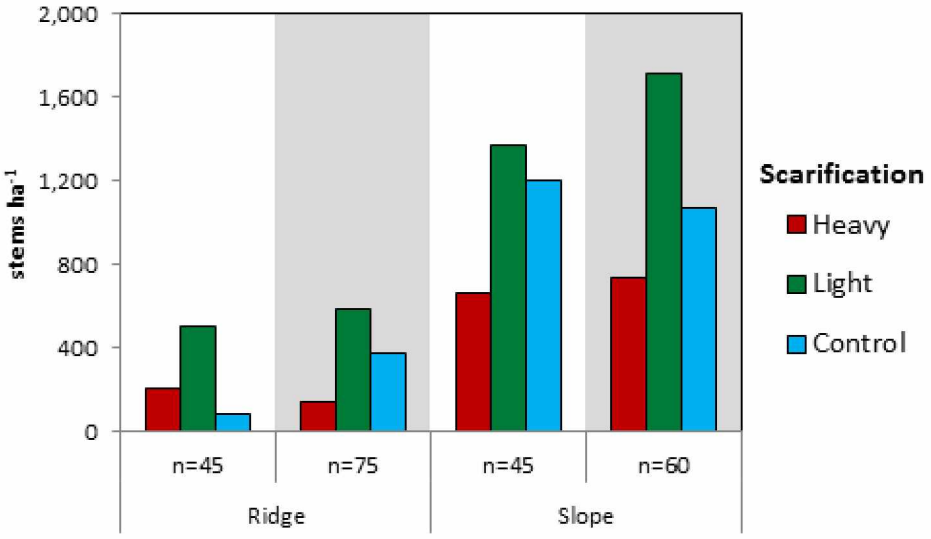

Figure C.4: Aspen stem density by scarification category and sample size. High aspen variability and non-significance of white spruce stocking method indicate that aspen response to scarification may be clarified by expanding sample size to include additional S/SPs sampled for less common white spruce stocking methods. Note that effects of scarification become more similar between LF whole plot with larger sample size.